

SAB Draft Report to Assist Meeting Deliberations -- Do not Cite or Quote

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FIRST EXTERNAL REVIEW DRAFT

PREPARED BY THE

USEPA SCIENCE ADVISORY BOARD'S

INTEGRATED NITROGEN COMMITTEE

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Chapter 1: Executive Summary

Introduction

Reactive nitrogen (Nr) encompasses biologically active, chemically reactive, and radiatively active nitrogen compounds. At the global scale, human activities now create ~2-fold more Nr than natural terrestrial ecosystems produce. The activities include the production of Nr as NH₃ as by the Haber-Bosch process for artificial fertilizer and as an industrial feedstock, the enhancement of biological nitrogen fixation (BNF) by crop cultivation (e.g., legumes) and the combustion of fossil fuels. The first two form Nr on purpose; the last one forms Nr as a byproduct, by accident.

There are large-scale impacts due to the creation of this additional Nr by humans. The first and foremost is the production of food. Without the creation of Nr for use as a fertilizer, the world population would be ~50% less than current levels.

However, for a number of reasons, essentially all of the Nr created by humans is lost to the environment where it circulates between, and accumulates in, environmental reservoirs. Once lost to the environment the Nr contributes to a number of adverse environmental effects, including photochemical smog, increased levels of N-containing aerosols, decreased atmospheric visibility, acid deposition, coastal eutrophication, greenhouse effect 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;
2. Evaluate the contribution an integrated nitrogen 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 nitrogen research to support risk reduction.

The rest of this summary gives an overview of the Nr inputs to the US, the fate of the Nr in the US. The chapter then summarizes 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.

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1 **Nr in the US: Inputs, Cycling, Impacts and Integrated Risk Reduction Strategies**

2 **Nr Inputs to US**

3 At the global scale, humans introduce ~2-fold more Nr than do natural terrestrial processes; in
4 the US it is ~5-fold. Natural ecosystems in the US introduce about 6.4 Tg N per year. In contrast,
5 human activity results in the introduction of about 29 Tg N per year (Figure 1-2). Fossil fuel
6 combustion introduces 3.8 Tg N/yr from transportation sources and 1.9 Tg N/yr from stationary
7 (utility and industry) sources, almost entirely as NO_x. It introduces ~5.7 Tg N per year into the
8 environment (combustion of wood and other forms of biomass generally occurs at temperatures
9 too low to convert N₂ to Nr).

10

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

15

16 A third source of Nr introduced into the US is cultivation-induced BNF, which introduced 7.7 Tg
17 N/yr. A small amount of Nr is also imported in grain and meat; in 2002 it was ~0.2 Tg N.

18

19 In summary, the largest sources of Nr created by human action in the USA are food production
20 and fossil fuel combustion. Although fossil fuel combustion is widely recognized within EPA
21 and society in general to be a major source of nitrogen, sulfur, and carbon pollutants and
22 resulting environmental quality concerns, in fact, food production and consumption are much
23 larger (about four times larger!) sources of reactive nitrogen than fossil fuel combustion.

24

25 **Nr Cycling and Fate in the US**

26 There are several possible fates for the ~35 Tg N of Nr that is introduced into the US. Emission
27 of N₂O removes 0.8 Tg N into the global atmosphere. Of the 6.3 Tg N of NO_x emissions, 2.7 Tg
28 N are deposited back to the US, and by difference we estimate that 3.6 Tg N are advected out of
29 the US atmosphere; similarly, of the 3.1 Tg N of NH₃ that are emitted to the US atmosphere, 2.1
30 Tg N are re-deposited, and 1 Tg N is advected via the atmosphere. Riverine injection of Nr to
31 the coastal zone accounts for 4.8 Tg N, while export of N containing commodities (e.g., grain)
32 removes another 4.3 Tg N from the US. All total, these losses sum to 14 Tg N, leaving 21 Tg N
33 unaccounted for. Of this amount, we estimate that 5 Tg N are stored in soils/vegetation and
34 groundwater, and, by difference, we estimate that 16 Tg N are denitrified to N₂ (Figure 1-2).
35 There are substantial uncertainties (+/- 50%) for some of these terms, especially those that
36 involve NH_x emission and deposition, and all the terms that are arrived at by difference (e.g.,
37 atmospheric advection; denitrification). These uncertainties drive the first tier of
38 recommendations of this report.

39 **Consequences, Impacts and Metrics for Nr**

40 The best and most important consequence of Nr is food production in the US, and global food
41 security. There are however, numerous negative consequences from anthropogenic Nr, including
42 photochemical smog, atmospheric particulate loading, ecosystem fertilization, acidification,
43 and/or eutrophication, greenhouse effect and stratospheric ozone depletion. But mitigating risk

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1 from these factors is difficult because one reactive N-containing molecule can contribute to all of
2 these effects as a consequence of the nitrogen cascade (Figure 1-1). Nitrogen is a dynamic
3 element easily transformed from one species to another and is transported rapidly through and
4 between ecosystem reservoirs. These characteristics make it an especially challenging element
5 to control.

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

9 Various approaches can be used to prevent, eliminate, reduce, or otherwise manage risk.
10 Understanding the environmental impacts of Nr can inform decisions on how best to manage
11 nitrogen risks. There are two main approaches to this problem – traditional impacts and
12 ecosystem services.

13 Traditional impacts include global warming, eutrophication, ecotoxicity, human health (cancer
14 and non-cancer), acidification, smog formation, and ozone depletion, among others. Sometimes
15 these impacts can be expressed in collective metrics. Collective metrics have the considerable
16 advantage of defining a straightforward framework within which environmental standards can be
17 derived that are protective of human health and the environment, the principal mission of the
18 USEPA. Such metrics also encourage evaluation of damage from collective sources, as long as
19 the characterization metric used is genuinely representative of the impact of a given contaminant.
20 Thus, for example, the total impact of acidic gases such as SO₂ and NO_x on the acidification of
21 watersheds can be expressed as a common metric. However, metrics for human health are
22 generally not as simple to characterize nor are there defined end points, thus the mechanism of
23 toxicity, number of individuals affected, value of lost workdays, medical treatment costs, and
24 value of human lives lost may all be used.

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

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

38 *Tradeoffs Among Nr Risk Reduction Options are Complex*

39 Once the foreseeable impacts are understood and the suite of benefits associated with various
40 risk reduction options described, then managers can consider trade-offs. Risk reduction
41 integration provides an intellectual framework that allows managers to make informed decisions

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1 about which benefits may need to be relinquished for other benefits when not all the desired
2 benefits can be achieved. For example, limiting nitrogen fertilizer application to reduce risks
3 from Nr applied to agro-ecosystems risks reduced yields and higher commodity prices, which in
4 turn may result in expansion of crop production area at the expense of natural wetlands,
5 grasslands, and forests.

6 *Measurement of Nitrogen in the Environment*

7 What you measure determines both what you do and how you gauge success or failure. Most
8 regulations set limits or specify control technologies for specific forms of nitrogen without
9 regard to the ways in which nitrogen is transformed once introduced into the environment.
10 Normally regulations also require some form of monitoring to document compliance.
11 Monitoring of these specific forms of nitrogen is not enough. There is a need to measure,
12 compute, and report the total amount of Nr, in appropriate units, present in impacted systems in
13 appropriate units because one form of Nr can be quickly converted to other forms.

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

20 In order to determine the extent of damage caused by excess N in environmental reservoirs, one
21 needs to know the Nr concentration or loading within a reservoir, and the threshold at which
22 negative impacts are manifested. This then provides a target that can be used to guide strategies
23 to decrease the Nr in the reservoir. The thresholds for impacts are better known for some
24 impacts relative to others. For example, the impacts of ozone on human health are known well
25 enough to have a standard set for both ozone, and for NO_x, an ozone precursor. The same can be
26 said for the impacts of Nr discharge to coastal waters—TMDLs are used to link loading to
27 impact. On the other hand, the impacts of Nr deposition on ecosystems, is only generally known.
28 There is strong scientific evidence to show that N deposition rates of 10 – 20 kg N/ha/yr can
29 cause negative impacts on a variety of ecosystems. However, there a large part of the land
30 surface in the northern hemisphere receives deposition receives N deposition in that range, it is
31 necessary to better define the link between N deposition and ecosystem response. Further, and
32 related to the previous section, our knowledge of N deposition is uncertain, especially for the
33 reduced N species, inorganic and organic. This knowledge needs to be improved to better link
34 deposition to ecosystem response (see Recommendation xx)

35 **Integrated Risk Reduction Strategies for Nr**

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

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1 *Control Strategies for Nr*

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

- 4 1. Transformation—in which one form of nitrogen is converted to another form (e.g.
5 nitrification of wastewater, denitrification),
6
- 7 2. Removal—in which Nr is sequestered from impacting a particular resource (e.g.
8 ion exchange)
9
- 10 3. Source limitation—in which the amount of Nr introduced into the environment is
11 lowered (e.g. lower fertilizer application rates, controls on NO_x generation)
12
- 13 4. Improved use efficiency—in which the efficiency of production that is dependent
14 on Nr is improved (e.g. increased grain yields for lower Nr applied, or reduced NO_x
15 from more efficient energy sources)
16
- 17 5. Improved practices—in which the flux of Nr that creates an impact is lowered
18 through better management practices (e.g. on-field agricultural practices, control of
19 urban runoff, controlled combustion conditions)
20
- 21 6. Product substitution—in which a product is developed or promoted which has a
22 lower dependency on Nr (e.g. switchgrass instead of corn grain as a feedstock for
23 ethanol)
24

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

27 *Management of Nr in the Environment*

28 Generally speaking, US environmental policy employs four mechanisms for the management of
29 contaminants in the environment:

- 30 1. Command-and-Control—in which permitted limitations on emissions, as promulgated
31 under various statutes, are issued. Violations may result in the assessment of penalties.
32
- 33 2. Government-based programs for effecting a policy, such as directed taxes, price supports
34 for a given commodity, subsidies to bring about a particular end, and grants for capital
35 expansion or improvement.
36
- 37 3. Market-based instruments for pollution control in which cap and trade markets are used
38 to bring about a desired policy end, often at reduced overall cost.
39
- 40 4. Voluntary programs in which desired ends are achieved using private or government-
41 initiated agreements or through outreach and education.
42

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1 An integrated approach to the management of Nr must of necessity use a combination of
2 mechanisms, each most appropriate to the nature of the problem at hand, that are supported by
3 critical research on reducing the risks of Nr, and reflective of an integrated policy that recognizes
4 the complexities and tradeoffs associated with the nitrogen cascade. Control at one point in the
5 cascade may be more efficient and cost effective than control or intervention at another point.
6 This is why understanding the nature and dynamics of the N cascade is so critically important.

7 **Findings and Recommendations**

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

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

15 There is over-lap among these groups as some recommendations and some are easier to do than
16 others. But collectively they represent an integrated view of what is needed to develop an
17 integrated Nr management strategy, based on sound science, and keeping in mind the nutritional
18 demands of the US and its trading partners. The specific recommendations are placed in the
19 relevant sections below.

20 In addition to providing these specific recommendations, we also make these four over-arching
21 recommendations that transcend the specific foci of specific sections.

22 OR 1-1. We recommend that EPA pursue an integrated approach to develop the
23 understanding necessary for science-based policies, regulations, and incentives to avoid
24 and remediate the impacts of excess Nr on the environmental, human health, and climate.
25 Such integration must cut across media (air, land, and water), Nr form (oxidized and
26 reduced), federal agencies, and existing legislative statutes (e.g., EISA, the Clean Air
27 Act, and the Water Quality Act).

28
29 OR 1-2. We recommend that the EPA form an *Intra-agency Task Force on Managing Nr*
30 that builds upon existing Nr efforts within the Agency, with the main purpose being to
31 identify the most cost-effective approaches to avoid the negative impacts of Nr loads
32 cascading through the environment because it poses a significant threat to human health
33 and environmental quality, and is a powerful driver of climate change.

34
35 OR 1-3. We recommend that an *Inter-agency Task Force on Managing Nr* be formed, with
36 EPA as the lead agency that includes at a minimum USDA, DOE, DOT, NOAA, and
37 USGS. The responsibility of this Task Force is to coordinate federal programs that
38 address Nr concerns and help ensure clear leadership roles for specific functions in
39 monitoring, modeling, researching, and regulating Nr in the environment.
40

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1 OR 1-4. We recommend that the purview of these task forces includes the following
2 elements

- 3 1) Research and science in support of integrated Nr management that includes
4 evaluation of critical loads; Nr budgets and life cycle accounting; monitoring as
5 the basis for informed policies, regulations, and incentive frameworks for
6 addressing excess Nr loads; the development and use of systemic models for Nr
7 management; new technologies; fertilizer and nutrient BMPs; development of Nr
8 indicators necessary for the assessment of effects related to excess Nr on human
9 health and the environment; combined C and N effects; indicators/endpoints,
10 costs, benefits and risks associated with the impairment of human health and
11 decline and restoration of ecosystem services). Research under this
12 recommendation should emphasize a systems approach.
13
- 14 2) The need for new regulations (and acceleration/extension of existing regs) and
15 standards (e.g. a means of accomplishing the goals of CAIR, acceleration of
16 compliance of uncontrolled EGUs).
17
- 18 3) New indicators (e.g. combined NO_y/NH_x, ecological responses)
19
- 20 4) Education, outreach, and communication
21
- 22 5) Economic incentives, particularly those that integrate air, aquatic, and land
23 sources of Nr (markets, taxes, subsidies)
24
- 25 6) New infrastructures (e.g. stormwater control, treatment of Nr point sources)
26
- 27 7) Review of enabling legislation for purposes of extending regulatory authority or
28 streamlining procedures for enacting Nr risk reduction strategies.
29

30 In addition to these overarching recommendations, the committee highlights the following
31 Recommendations that collectively would result in ~25% decrease in the amount of Nr lost to
32 the US environment.

33 **Recommendation R1-1:** We suggest a goal of decreasing livestock-derived ammonia emissions
34 to approximately 80% of 1990 emissions, a decrease of **0.5 Tg N** per year (by a combination of
35 Best Management Practices and engineered solutions). This will reduce PM_{2.5} by ~0.3 µg/m³
36 (2.5%) and improve health of ecosystems by achieving progress towards critical load
37 recommendations. Additionally we recommend decreasing ammonia emissions derived from
38 fertilizer applications by 20% (decrease by ~**0.2 Tg N** per year.).

39 **Recommendation R1-2:** We recommend that excess flows of Nr into streams, rivers, and
40 coastal systems be decreased by approximately 20% (~**1 Tg N** per year.) through improved
41 landscape management without undue disruption to agricultural production and human lifestyles
42 and economies. This would include activities such as using wetland management (e.g., USDA
43 Wetlands Protection Program), improved tile-drainage systems and riparian buffers on cropland,

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1 and implementing storm water and non-point source management practices (e.g., EPA permitting
2 and funding programs).

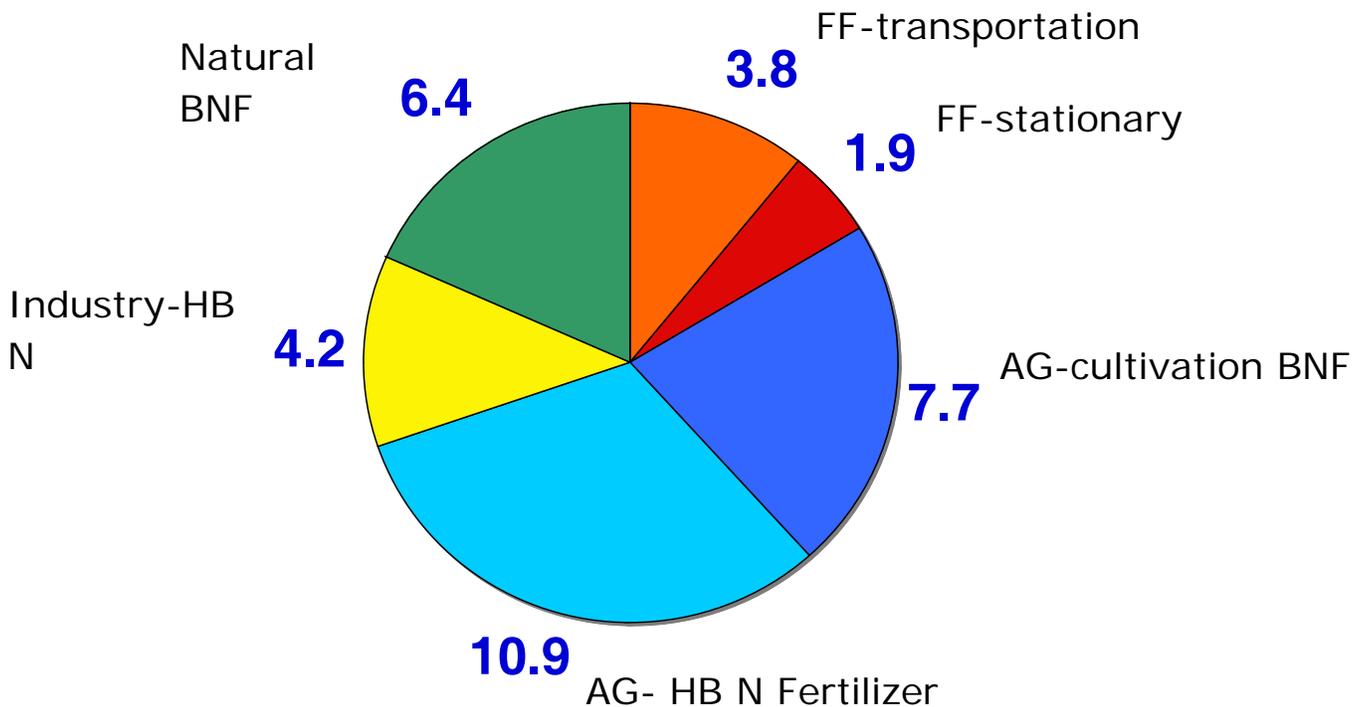
3 **Recommendation R1-3:** We recommend that crop output be increased while reducing total Nr
4 by up to 20% of applied artificial Nr. This action would result in a savings of ~2.4 Tg N per year,
5 below current levels of Nr additions to the environment.

6 **Recommendation R1-4:** We recommend that a high priority be assigned to nutrient
7 management through a targeted construction grants program under the CWA. The committee
8 believes that **0.5 to 0.8** Tg N per year can be saved from Nr inputs to the environment.

9 In summary, if optimally effective and well-integrated management strategies for reactive
10 nitrogen are to be developed and implemented, it is critical to understand the relationship among
11 various ecological risks and food-production and energy-production benefits and how much of
12 the newly created reactive nitrogen is transferred among various compartments, as well as the
13 effects excess Nr have on both human health and human welfare and the ecosystems on which
14 the quality of life depends.

15

16 *Figure 1-1: New Nr introduced into the US, 2002, Tg N.*

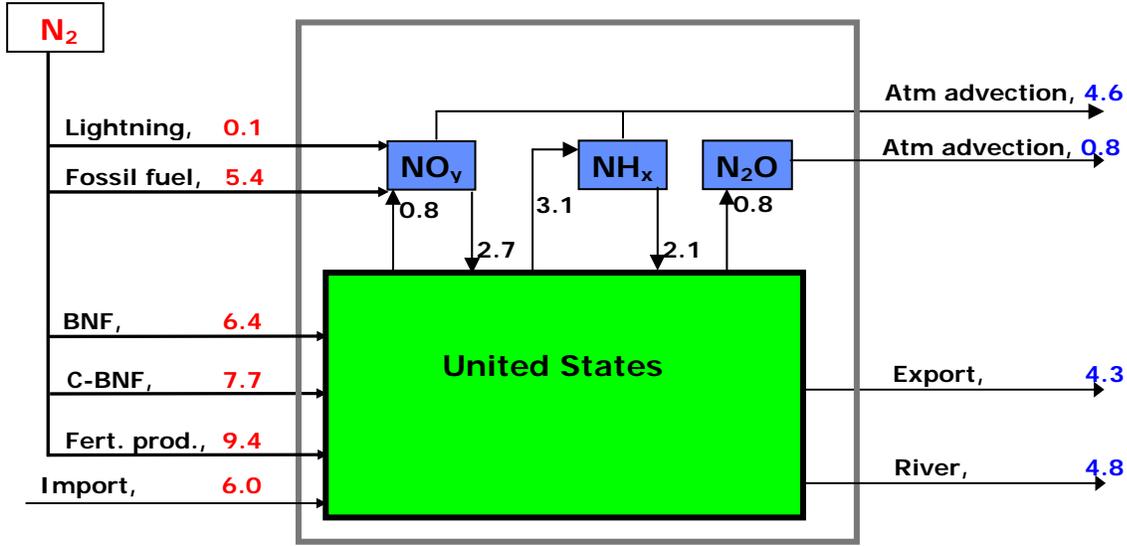


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**US Nitrogen Budget
Tg N yr-1**



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 N₂:
21 Tg N - 5 Tg N = 16 Tg N

Figure 1- 2: US N Cycle

Chapter 2: Introduction

2.1 General Background About Environmental Impacts of Nitrogen 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 nitrogen. And just as most of the water on the planet is not useable by most organisms, most of the nitrogen is also unavailable.

Approximately 78% of the atmosphere is diatomic nitrogen (N₂), which is unavailable to most organisms because of the strength of the triple bond that holds the two nitrogen atoms together. Over evolutionary history, only a limited number of species of Bacteria and Archaea have evolved the ability to convert N₂ to reactive nitrogen (Nr¹) via biological nitrogen fixation. However, even with adaptations to use nitrogen efficiently, many ecosystems of the world are limited by nitrogen.

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

This deficiency led to what arguably has been called one of the world's most important discoveries—how to extract N₂ from the atmosphere and convert it to NH₃—the Haber-Bosch process (Erisman et al., 2008). Today this process and cultivation-induced biological nitrogen fixation (C-BNF) introduce over 140 Tg N yr⁻¹ of Nr into the global environment to sustain food production. Another 23 Tg N yr⁻¹ are introduced by the Haber-Bosch process for the chemical industry, and 25 Tg N yr⁻¹ are introduced via the combustion of fossil fuels.

The total global anthropogenic Nr creation rate is ~190 Tg yr⁻¹ (2005), substantially larger than the median of estimates for Nr creation by natural terrestrial processes (~100 Tg N yr⁻¹) (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,

¹ The term reactive nitrogen (Nr) as used in this paper includes all biologically active, chemically reactive, and radiatively active N compounds in the atmosphere and biosphere of Earth. Thus, Nr includes inorganic reduced forms of N (e.g., NH₃ and NH₄⁺), inorganic oxidized forms (e.g., NO_x, HNO₃, N₂O, and NO₃⁻), and organic compounds (e.g., urea, amines, and proteins), by contrast to unreactive N₂ gas.

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1 with global commodity stocks running at a 58 day supply and food prices increasing
2 dramatically, the challenge is to increase the nutrient use efficiency of Nr in agricultural
3 systems while maintaining or increasing yields. (USDA ERS/World Agricultural Outlook
4 Board, July 11, 2008. World Agricultural Supply and Demand Estimates)
5

6 There are large regional disparities in Nr creation rates on both absolute and per capita bases.
7 Total Nr creation is larger in Asia than in any other region. Per capita Nr creation is largest in
8 North America and Europe. Humans also redistribute large amounts of Nr among countries or
9 regions of the world through exports of fertilizers, feed grains, and fossil fuels. Nevertheless,
10 there are large regions of the world with populations approaching 1 billion, where there is
11 malnutrition due to a lack of adequate supply of available Nr to sustain crop production, among
12 other reasons.
13

14 The introduction of Nr into most regions of the USA by humans has greatly increased food
15 availability. However, since essentially all the Nr created for food production and by fossil
16 fuel combustion is lost to the environment, it has also greatly increased Nr's contribution to a
17 wide variety of environmental issues. Most plants, animals, and microorganisms are adapted
18 to efficiently use and retain small increments of additional Nr. Addition of Nr to most
19 ecosystems may first lead to increased uptake, growth, storage, and hence to increased
20 biomass, including food or fiber production. However, further addition of Nr in excessive
21 amounts often leads to imbalances in the N cycle and potential leakages in the form of air
22 emission or water discharges into other ecosystems where it may disrupt ecosystem functions
23 and have a negative impact on resources. In essence, the assimilative capacity of the
24 ecosystem may be insufficient to benefit from increases in Nr without disruptive change.
25 While there will always be Nr losses during food production, the challenge is how to minimize
26 those losses while meeting the demand for food production.
27

28 Negative consequences of Nr injection into the US environment include increases in
29 photochemical smog and PM-2.5, decreases in atmospheric visibility, both increases and
30 decrease in productivity of grasslands and forests, acidification of soils and freshwaters,
31 accelerating estuarine and coastal eutrophication, increases in the emission of greenhouse gases
32 to the atmosphere and decreases in stratospheric ozone concentrations. All of these changes in
33 environmental conditions lead to a variety of negative impacts on both ecosystem and human
34 health. These changes impact air, land, water and the balance of life in an interrelated fashion
35 are often referred to a cascade of effects from excess Nr, or the "nitrogen cascade" (Figure 2-
36 1). Unlike other element-based pollution problems, the nitrogen cascade links the negative
37 impacts, where one N-containing molecule can in sequence contribute to all the environmental
38 issues mentioned above.
39

40 The nitrogen cascade comprises, and has consequences for three components:
41

- 42 1. biogeochemical,
 - 43 2. environmental changes and
 - 44 3. human and ecosystem consequences (Figure 2-1).
- 45

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1 The “biogeochemical” component of the nitrogen cascade include Nr creation from N₂ as a
2 consequence of chemical, food and energy production, Nr use in food and chemical
3 production, Nr losses to the environment, changes in Nr species residence times in
4 environmental reservoirs, Nr transfers among reservoirs and Nr conversion back to N₂. The
5 “environmental changes” component arises from the fact that increased Nr levels in the
6 environment contribute to the issues noted above. The “consequences” component includes
7 negative ecosystem and human health impacts at local, regional, national and global scales.
8 The duality of nitrogen being a critical resource but also a contributor to many of the
9 environmental concerns facing the USA today, makes it imperative to not only understand how
10 human action has altered N cycling in the USA, but also the consequences of those alterations
11 on people and ecosystems. The over-arching question is how do we protect and sustain an
12 ecosystem that provides multiple benefits to society while also providing the interconnected
13 material, food and energy required by society?
14

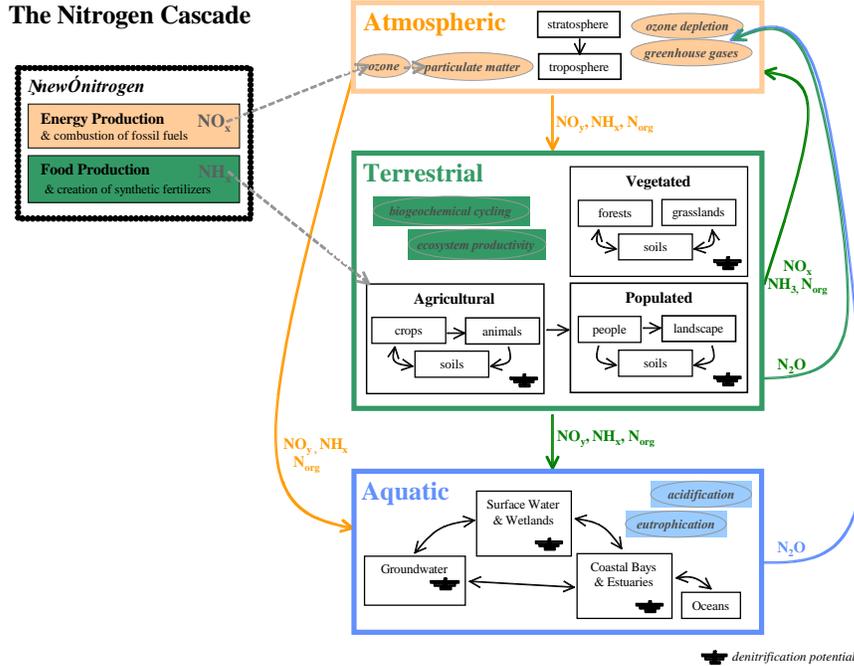
15 The principal mechanisms for Nr removal from circulation in the environment are complete
16 denitrification (re-conversion of Nr back to non-reactive gaseous N₂), and storage in long-term
17 reservoirs (e.g., soils, sediments, and woody biomass). In some cases, it may be possible to
18 capture Nr emissions or discharges and deliver them to food or fiber production areas where
19 there are nitrogen deficiencies. However, major challenges in the management of the N cycle
20 are how to decrease creation of Nr while still meeting societal needs, promote denitrification of
21 excess Nr (without producing N₂O), and improve the efficiency of use and reuse of excess Nr
22 in a cost-effective manner. Solving both these challenges will result in less Nr accumulation.
23

24 In summary, Nr inputs to the nation and the world have been increasing, largely due to human
25 activities associated with food production and fossil fuel combustion. Despite the obvious
26 benefits of a plentiful supply of food and energy, the adverse consequences associated with the
27 accumulation of Nr in the environment are large, with implications for human health and the
28 environment.
29

30 The greater the inputs of Nr to the landscape, the greater the potential for negative effects,
31 caused by greenhouse gas production, ground level ozone, acid deposition, and Nr overload
32 that can contribute to climate change, degradation of soils and vegetation, acidification of
33 streams, lakes and rivers, estuarine and coastal eutrophication, hypoxia and habitat loss.
34

35 The growing nature of the Nr problem, and the adverse and intertwined consequences
36 associated with Nr inputs to air, land, and water as exhibited in the nitrogen cascade
37 underscore the need for researchers and managers to explore integrated strategies that
38 minimize N inputs, maximize its use efficiency, promote Nr removal processes and protect
39 humans and natural resources.
40
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7 *Figure 2- 1: The Nitrogen Cascade: The popular concept of the nitrogen cascade highlights*
 8 *that once a new Nr molecule is created, it can, in sequence, travel throughout the environment*
 9 *contributing to major environmental problems (Galloway et al., 2003). This adaptation of the*
 10 *cascade was developed by the Integrated Nitrogen Committee to provide a context for*
 11 *considering nitrogen-related issues and ecosystem effects in the USA. To consider the*
 12 *cascading effects of Nr in the USA, we examine the relative sizes of the various Atmospheric,*
 13 *Terrestrial, and Aquatic compartments where Nr is stored, and the magnitudes of the various*
 14 *flows of nitrogen to, from, and within them. The nitrogen cascade concept implies the cycling*
 15 *of Nr among these compartments. The important process of denitrification is the only*
 16 *mechanism by which Nr is converted to chemically inert N₂, ‘closing’ the continuous cycle.*

17

18 *The “new” nitrogen box depicts the two primary anthropogenic sources by which Nr*
 19 *originates, energy production and food production, and where they enter ecosystems. Energy*
 20 *production includes both fossil fuel and biofuel combustion. Food production includes N*
 21 *fertilizer produced in the USA, cultivation-induced biological nitrogen fixation in the USA,*
 22 *production of animals and crops in the USA for human consumption, and imports of N-*
 23 *containing fertilizer, grain and meat to the USA.*

24

25 *The Atmospheric compartment indicates that tropospheric concentrations of both ozone and*
 26 *particulate matter are increased due to NO_x emissions to the atmosphere. The ovals illustrate*
 27 *that the increase in N₂O concentrations, in turn, contribute to the greenhouse effect in the*
 28 *troposphere and to ozone depletion in the stratosphere. Except for N₂O, there is limited Nr*

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1 *storage in the atmosphere. Losses of Nr from the Atmospheric compartment include NO_y,*
2 *NH_x, and Norg deposition to Terrestrial and Aquatic ecosystems of the earth's surface. There*
3 *is little potential for conversion of Nr to N₂ via denitrification in air. However, once airborne*
4 *deposition of Nr occurs it will be subject to denitrification pathways via soil and water.*

5
6 *The Terrestrial compartment depicts that Nr enters agricultural lands via food production and*
7 *is introduced to the entire terrestrial landscape via atmospheric deposition. Within*
8 *'agricultural' regions there is cycling between soils, crops and animals, and then a transfer of*
9 *Nr as food to 'populated' regions, from which there are Nr losses (e.g, sewage, landfills). The*
10 *ovals showing 'ecosystem productivity' and 'biogeochemical cycling' reflect that Nr is actively*
11 *transported and transformed within the Terrestrial compartment, and that as a consequence*
12 *there are significant impacts on ecosystem productivity due to fertilization and acidification,*
13 *often with resulting losses of biodiversity. There is ample opportunity for Nr storage in both*
14 *biomass and soils. Losses of Nr from this compartment occur by leaching of NO_x, NH₃, and*
15 *Norg to Aquatic ecosystems and by emissions to Atmospheric compartment as NO_x, NH₃,*
16 *Norg, and N₂O. There is potential for conversion of Nr to N₂ via denitrification in the*
17 *Terrestrial compartment.*

18
19 *The Aquatic compartment shows that Nr is introduced via leaching from Terrestrial*
20 *ecosystems and via deposition from Atmospheric ecosystems. Connected with the hydrological*
21 *cycle, there are Nr fluxes downstream with ultimate transport to coastal systems. Within the*
22 *Aquatic compartment, the ovals highlight two significant impacts of waterborne Nr—*
23 *acidification of freshwaters and eutrophication of coastal waters. Except for Nr accumulation*
24 *in groundwater reservoirs, there is limited Nr storage within the hydrosphere. Losses of Nr*
25 *from the Aquatic compartment are primarily via N₂O emissions to the Atmospheric*
26 *compartment. There is a very large potential for conversion of Nr to N₂ via denitrification in*
27 *water and wetlands.*

28
29 **2.2 Overview of EPA Research and Risk Management Programs in Context of Other**
30 **Research and Management Programs**

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

- 34 1. Clean Air and Global Climate Change,
- 35 2. Clean and Safe Water,
- 36 3. Land Preservation and Restoration,
- 37 4. Healthy Communities and Ecosystems, and
- 38 5. Compliance and Environmental Stewardship.

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

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

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

16 The Office of Research and Development's mission is to conduct leading-edge research
17 and foster the sound use of science and technology in support of EPA's mission. ORD is
18 well recognized for providing a scientific basis for the development of the NAAQS
19 standards for NOx and particulate matter. ORD's revised Multi-Year Plan for Ecological
20 Research will identify and quantify ecological services affected by nitrogen pollution and
21 controls across spatial scale and media and in combination with place-based and
22 ecosystem-specific (wetlands) research.

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

31 Out of a proposed three trillion federal budget, almost one trillion goes to "Discretionary
32 Programs" of which EPA is one. EPA requested \$7.14 billion for FY2009. The requests
33 for some other agencies with programs relating to nitrogen risk reduction or research are:

34	Agriculture	\$ 20.8 billion
35	Commerce	\$ 8.2 billion
36	Defense	\$515.4 billion
37	Energy	\$ 25.0 billion
38	Interior	\$ 10.6 billion

39

40 The budgets for these departments have increased since FY 2001; EPA's has decreased.

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1 EPA's budget is not funded or organized by individual pollutants, so it is difficult to say
2 how much of this supports reducing risks from reactive nitrogen.

3 The proposed research and development budget for the federal government for FY2009 is
4 about \$145 billion. The portion of this budget defined as environmental portion is a little
5 over two billion mostly shared by EPA, National Oceanographic and Atmospheric
6 Administration in the Department of Commerce, the U.S. Geological Survey in the
7 Department of Interior, the Corps of Engineers, and the Forest Service in the Department of
8 Agriculture. Of the two billion, EPA's research program receives about \$450 million.
9 Enough environmentally-related work is done under other research missions at the
10 Department of Agriculture, the Department of Energy, the National Institute of
11 Environmental Health Science, the National Aeronautics and Space Administration, etc
12 that federally funded environmental research and development totals about eight billion.

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

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

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

24 **2.3 The Need for Integration**

25
26 Some impacts of nitrogen on ecosystems and people have been known for centuries (e.g.,
27 impacts of atmospheric deposition on agricultural crops); others for only a few decades (e.g.,
28 impact of N₂O on stratospheric ozone) and still others are still being debated (e.g., impact of
29 ingested Nr on human health). Notwithstanding current uncertainties, the EPA and its
30 predecessor organizations have been active in the management of Nr for a variety of reasons,
31 including decrease in the Nr amount in sewage, control of NO_x to decrease photochemical
32 smog and acid rain, control of Nr inputs to coastal systems, controls on fine particulates in the
33 atmosphere and decrease in Nr leaching from crop and animal production systems. As
34 beneficial as those efforts have been, they focus on the specific problem without consideration
35 of the interaction of their particular system with other systems downstream or downwind.
36 Given the reality of the nitrogen cascade, this approach may result in short term benefits for a
37 particular system but will also likely only temporarily delay larger scale impacts on other
38 systems. Thus there is a need to integrate nitrogen management programs, to ensure that
39 efforts to lessen the problems caused by nitrogen in one area of the environment do not result
40 in unintended problems in other areas.

41

1 **2.4 Charge and Scope of SAB INC Report**

2
3 In 1973 the Science Advisory Board issued its first report, the 200-page *Nitrogenous*
4 *Compounds in the Environment*. The report addressed sources and effects of nitrogenous
5 compounds, including those from air emissions, animal wastes, crop agriculture, industrial
6 processes, and solid wastes. After concluding that, “At present, all known trends appear to
7 be ones that can be managed and kept within control, if appropriate steps are taken now,”
8 the SAB provided recommendations relating to research on and control of reactive
9 nitrogen. Later, the SAB would consider 1970s to be the first step in environmental
10 protection, characterized by broad agreement on environmental problems and their sources.
11 The second step emerged in the 1980s when the risk assessment/risk management
12 paradigm proposed by the National Research Council in 1983 achieved wide acceptance.

13
14 In *Toward Integrated Environmental Decision-Making*, published in 2000, the SAB
15 articulated a third step in environmental protection -- the Framework for Integrated
16 Environmental Decision-Making. In this 2000 report, the SAB noted that the 3-phase
17 structure (problem formulation, analysis & decision-making, followed by implementation
18 and evaluation), “belies the complexities involved in putting the concept of integrated
19 decision-making into practice”.

20
21 The SAB’s interests in nitrogen science and integrated environmental protection would
22 converge in 2003, when the SAB identified integrated nitrogen research and control
23 strategies as an important issue facing the Agency and formed the Integrated Nitrogen
24 Committee to undertake a study of this issue. To provide a basis for the advice and
25 recommendations, the Committee:

- 26
27 1. Identified and analyzed, from a scientific perspective, the problems nitrogen
28 presents in the environment and the links among them;
29
30 2. Evaluated the contribution an integrated nitrogen management strategy could make
31 to environmental protection;
32
33 3. Identified additional risk management options for EPA’s consideration; and
34
35 4. Made recommendations to EPA concerning improvements in nitrogen research to
36 support risk reduction.

37
38 In the course of its study, the Integrated Nitrogen Committee held four public face-to-face
39 meetings at which it invited briefings from EPA’s Office of Air and Radiation, Office of
40 International Affairs, Office of Research and Development, and Office of Research and
41 Development; from the Department of Agriculture’s Agricultural Research Service,
42 Cooperative State Research, Extension and Education Service, and the Economic Research
43 Service; and external organizations such as the Energy Research Centre of the Netherlands,
44 Environmental Defense Fund, International Plant Nutrition Institute, Iowa State University,
45 LiveFuels, and the Soil and Water Conservation Society.

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1 Additionally, the INC invited scientists and managers from EPA, other federal agencies,
2 states and localities, academia, non-governmental organizations and the private sector to
3 participate in its October 20-22, 2008 Workshop Meeting on Nitrogen Risk Management
4 Integration and requested external peer review comments from [TBD].
5
6

Chapter 3: Behavior of Reactive Nitrogen in the Environment

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

3.1 Introduction

Although nitrogen (N) is a major required nutrient that governs growth and reproduction of living organisms, losses of reactive nitrogen (Nr) 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 nitrogen oxides to the atmosphere via fossil fuel combustion. Consumption of food to meet nutritional requirements of a growing population results in agricultural emissions of ammonia, urban and industrial emissions of nitrogen oxides, and nitrous oxide as well as losses of nitrate 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 panel's work. The first two are the introduction of Nr into US systems from fossil fuel combustion and from food production (Section 3.2) 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 3.3). The third aspect is the impacts of Nr on humans and ecosystems (Section 3.4) 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, cultivation-induced biological nitrogen fixation 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 2-1). National level values for Nr fluxes are displayed in Table 3-1. 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 3-1. Reactive nitrogen fluxes for the USA, Tg N in 2002^a			
* Newly created reactive N			
Nr inputs to <i>Atmospheric</i> compartment		<u>Tg N/yr</u>	<u>%</u>
N ₂ O-N emissions		0.8	8
	agriculture - livestock (manure) N ₂ O-N	0.03	
	agriculture - Soil management N ₂ O-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 <i>Atmospheric</i> inputs	10.0	100
Nr inputs to <i>Terrestrial</i> compartment			
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 USA; *5.8 net imports to USA)			
	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 <i>Terrestrial</i> inputs	43.5	100
Nr inputs to <i>Aquatic</i> compartment			
surface water N flux		4.8	

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Table 3-1 Notes

- a. The Nr estimates in this table are shown with two significant digits or 0.1 Mmt N yr⁻¹ (or Tg N yr⁻¹) 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 (4 – 9) Tg N yr⁻¹ total anthropogenic Nr deposition to the entire 48 States (Section 3.3.1.10). The EPA sponsored CMAQ run yielded a value of 4.81 Tg N yr⁻¹. The value shown for the total (6.9 Tg N yr⁻¹) reflects the assumption that organo-nitrogen species should be added to the model estimate as 30% of the total.

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

Table 3-1 Data Sources:

- Emissions, N₂O-N (USEPA Inventory of US Greenhouse Gas Emissions and Sinks)
- Emissions, NH_x-N (USEPA National Emissions Inventory)
- Emissions, NO_x-N (USEPA 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 (USEPA CMAQ model)
- N₂ fixation in cultivated croplands (USDA census of agriculture, literature coefficients)
- N₂ 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)

3.2 Sources of Reactive Nitrogen

3.2.1 Nr sources to the US

We first consider the creation of “new” Nr in the environment; this refers to Nr that is either newly fixed within or transported into the United States, and highlights where Nr is introduced into ecosystems. New Nr arises from fossil fuel combustion, food production and materials production (Table 3-1).

Fossil fuel combustion emits Nr (mostly NO_x) to the atmosphere¹. 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 3-1). Another 0.2 Tg N/yr of NH₃-N and 0.1 Tg N/yr of N₂O-N is emitted from the same sources (Table 3-1). 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 are used primarily in agriculture to support food production. Nr is introduced into terrestrial landscapes of the US by production of fertilizers within the U.S. (9.4 Tg N/yr), and by net imports via world trade (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 3-1).

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 3-1).

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 3-1) 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.

¹ Nr is generally not formed during combustion of wood and modern biomass because of lower combustion temperatures.

3.2.2 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, nitrogen oxides are formed. The source of nitrogen is either the nitrogen contained in the fossil fuel or the nitrogen (N₂) that comprises about 80% of atmosphere. Fuel-derived nitrogen is important in the case of burning coal (which contains nitrogen) while atmospheric-derived nitrogen is formed during higher temperature processes that occur when gasoline or diesel fuel is burned in motor vehicles. NO_x is the predominant source of reactive nitrogen in these systems (Table 3-1). In the US, highway motor vehicles account for the largest manmade source of NO_x at 36% (Figure 3-1), while off-highway vehicles, electric utilities and industrial processes account for 22%, and 20%, respectively.

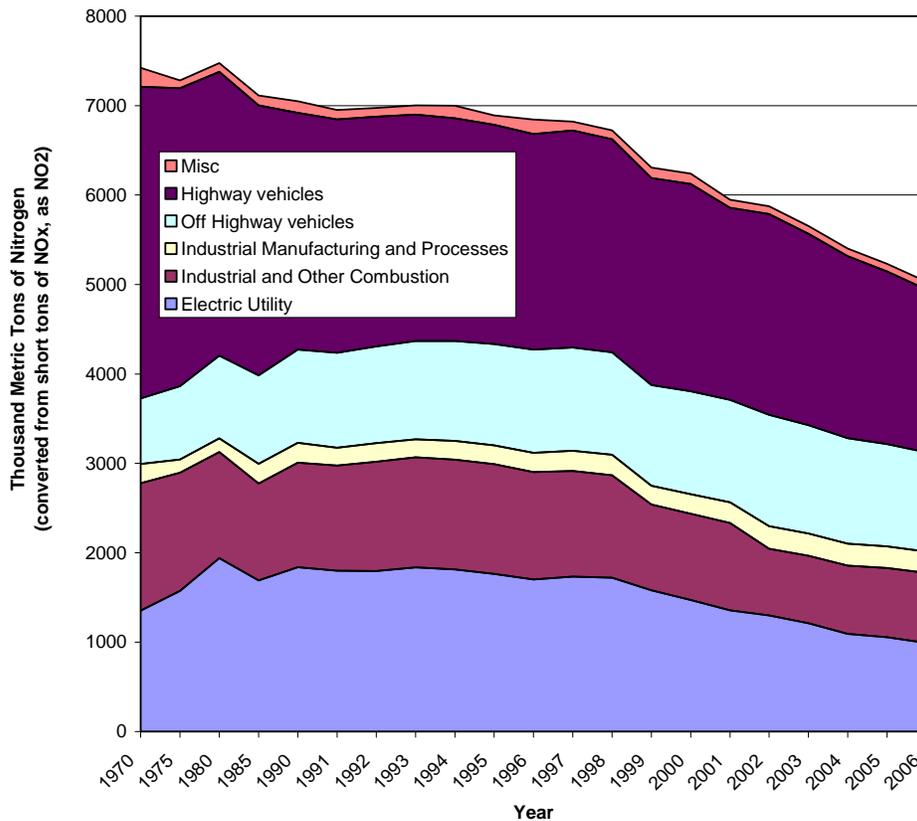


Figure 3-1. US NO_x emission trends, 1970-2006. Data are reported as thousand of metric tons of N converted from NO_x as NO₂.

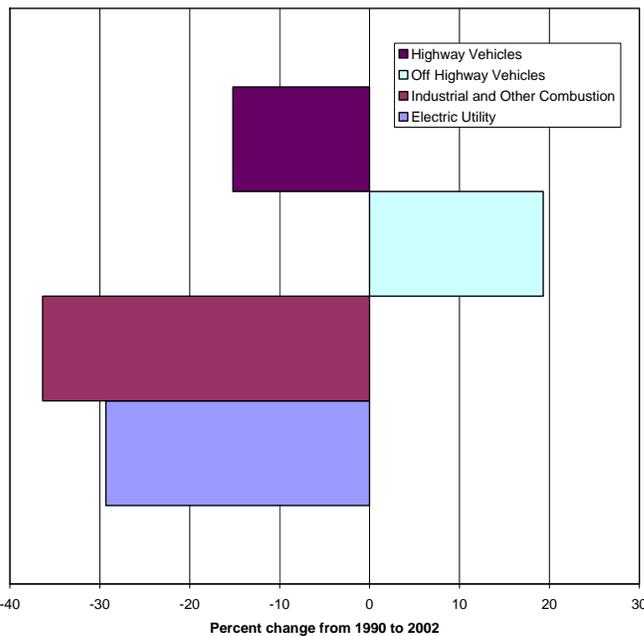
(Source: <http://www.epa.gov/ttn/chief/trends/index.html>)

Figure 3-1 also illustrates that the amount of nitrogen oxides (reported as metric tons of N) released from various fossil fuel sources has decreased dramatically from 1970. Total emissions were on the order of 7400 metric tons in 1970, reduced to 5900 in 2002, with

1 further reductions in 2006 to 5030 metric tons. Overall this represents a decrease of over
2 30%.

3
4 The top sources: highway vehicles, off-highway vehicles, electric utilities, and other
5 industrial and combustion systems show decreases between 15-30% (Figure 3-2).
6 Reductions were the highest for “other” systems followed by electric utilities. These
7 reductions are most likely the result of changes in regulations and control technologies
8 for these stationary systems. To a lesser extent, changes in highway vehicle regulations
9 and the removal of older fleets from the road has resulted in a decrease of approximately
10 15%; however, this reduction is accompanied by an increase in miles traveled, which
11 suggests that the actual decrease in a single vehicle is larger. Off highway vehicles
12 showed an increase in emissions, potentially due to better quantification of these sources.
13 Sources here include locomotives, marine, etc. While some regulations are in place for
14 some of these sources, such as locomotives, further control of these and other sources
15 could reduce emissions. In fact, technology development in the locomotive industry
16 shows that reductions of approximately 70% are possible. Further reductions would
17 require more innovative, expensive methods such as SCR with urea injection. Engine
18 manufacturers are also investigating using SCR systems for diesels. However, it must be
19 noted that these systems emit small amounts of ammonia and must be operated properly
20 to avoid trading off NO_x emissions for ammonia.

21



22
23 **Figure 3-2. Percent reductions in NO_x emissions, 1990-2002, from different sources**
24 **(off-road, on-road vehicles, power generation, etc)**

25
26 Texas, California, Florida, Ohio, and Illinois emissions, metric tons of nitrogen as
27 converted from tons of NO_x or NO₂) with the processes listed (Table 3-2), illustrate the
28 fact that individual emission scenarios are quite different.

29

	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

1
 2 **Table 3-2. Top 5 Emitters of N in metric tons (2001 data; based on tons of NO_x as**
 3 **NO₂) (Source: These data were derived from the 2001 information obtained at:**
 4 **<http://www.epa.gov/air/data/geosel.html>)**

5
 6 As seen in Table 3-2, Texas’ fuel combustion sources are on the same order as highway
 7 vehicles; this is in comparison to California, where vehicles, highway and off-highway
 8 are the dominate source (over 75%) for this state. These results are attributed to
 9 industries and coal-fired power plants located in Texas. Almost 40% of the power
 10 generation in Texas is due to coal-fired plants. On the other hand, California imports
 11 most of its coal-fired power and generates its own power predominantly from other
 12 sources, such as natural gas (50%), hydro and nuclear (33%). Louisiana and Texas have
 13 high emissions due to industry because of the chemical and oil industries located in these
 14 states. These results illustrate that many sources contribute to the NO_x emitted from
 15 energy sources and the number of automobiles is a factor. The number of automobiles is
 16 related to the population. The estimated population of California for 2006 is 36.4 million
 17 people versus Ohio and Illinois which are on the order of 11-12 million.

18
 19 **Finding F3-1:** While the overall emissions of NO_x have decreased, there are still some
 20 sources which have increased in emissions, namely off highway mobile sources. Some
 21 of these sources have little or no control technologies in place. In addition, there are a
 22 number of EGUs which have no NO_x control and even a larger number that do not have
 23 SCR technology.

24
 25 One off-highway source, locomotives, are under tiered emission limits, with Tier II to be
 26 in place from 2005-2011. To meet further limits (Tier IV, 2015 and later), innovative
 27 ultra-low emission engines are in place. In addition SCR systems are being developed
 28 for locomotives and, in some cases, diesel engines. These innovations are promising for
 29 further reductions. However, it should be noted, that reductions in NO_x should not come
 30 at the cost of increased NH₃ (currently a small fraction of the emissions, but a side
 31 reaction in SCR systems) or N₂O.

32
 33 Particularly with fossil combustion sources, future carbon management will link with
 34 nitrogen management. For example, using hybrid plug-ins reduces the amount of mobile

1 NOx emitted with little impact on the stationary source which still meets its emission
2 requirements. Increase in efficiency in combustion systems, which reduces carbon
3 dioxide emissions per unit of energy generated, will, in turn, reduce nitrogen.

4
5 **Recommendation R3-1:** In addition to current control efforts for power plants and
6 passenger cars, EPA should include other important unregulated mobile and stationary
7 sources, in particular off highway such as marine. Technology is in place to allow for
8 further reductions in many cases.

9 10 **3.2.3 Nr inputs and losses from crop agriculture**

11
12 Agriculture uses more Nr and accounts for more Nr losses to the environment than any
13 other economic sector. Synthetic fertilizers are the largest sources of Nr input to
14 agricultural systems. The next largest source is cultivation-induced BNF (Table 3-1).
15 The major pathways by which Nr is lost from these systems include nitrate losses from
16 leaching, runoff and erosion, and gaseous emissions via volatilization of ammonia and
17 nitrification/denitrification. Similar loss pathways occur for Nr that cycles through
18 livestock systems, which also account for a large portion of nitrogen flux (predominantly
19 as ammonia) in animal agricultural systems (Aneja et al. 2006). Therefore, assessment of
20 Nr impacts on the environment and development of strategies to minimize negative
21 impact must be based on a thorough understanding and accurate accounting of Nr fluxes
22 in both crop and livestock systems, and the trends in management practices that have
23 greatest influence on Nr losses from these systems (Aneja et al, 2008a,c).

24
25 In the past 60 years, nitrogen fertilizers have had a beneficial effect on agriculture both
26 nationally and globally by increasing crop yields. However, the high loading of Nr from
27 agricultural nutrient sources has led to deleterious effects on the environment, such as
28 decreased visibility from increased aerosol production and elevated nitrogen
29 concentration in the atmosphere, ground, and surface waters (Galloway et al. 2003).

30 31 *3.2.3.1 Nitrogen Fertilizer Use*

32
33 Obtaining accurate data on fertilizer use is a critical first step in understanding Nr cycles
34 in agriculture. There are several sources of data reporting fertilizer usage but it is not
35 clear whether data quality is sufficient for assessing environmental impact. Although the
36 Uniform Fertilizer Tonnage Reporting System (UFTRS) was developed to collect fees to
37 fund the consumer protection mission of State Chemists and fertilizer regulatory control
38 officials, it also provides data on fertilizer sales in many states, which in turn are used by
39 many agencies and environmental scientists to estimate consumption and use of
40 nitrogenous fertilizers in the US. The Association of American Plant Food Control
41 Officials (AAPFCO) tallies and publishes the statewide fertilizer sales data annually
42 (Terry et al. 2006), which is one of the most widely used sources of data on fertilizer use.
43 It is typically assumed that fertilizers are used in the same region in which they were
44 sold. The annual state-level data published by AAPFCO, which are based on commercial
45 fertilizer sales and often taxed at the state level (but not in all states), are the only data
46 source available, and it includes fertilizer sales for both agricultural and non-agricultural

1 purposes. These state-level data must then be allocated to counties, regions, or
2 watersheds in the states, and the algorithms used for this process are based on a number
3 of assumptions that address dealer/farmer storage, inventories, and cross-state sales
4 issues (personal communication, Stan Daberkow, USDA-ERS).

5
6 The USDA National Agricultural Statistics Service Information (NASS) fertilizer usage
7 data represents another source of information derived from farmer “agricultural chemical
8 use” surveys that provide information in six categories: field crops, fruits and vegetables,
9 nurseries/floriculture, livestock use, and post-harvest application. For each group, NASS
10 collects fertilizer, pesticide, and pest management data every year on a stratified random
11 sample of farmers at the field level

12 ([http://usda.mannlib.cornell.edu/usda/current/AgriChemUsFC/AgriChemUsFC-05-16-](http://usda.mannlib.cornell.edu/usda/current/AgriChemUsFC/AgriChemUsFC-05-16-2007_revision.pdf)
13 [2007_revision.pdf](http://usda.mannlib.cornell.edu/usda/current/AgriChemUsFC/AgriChemUsFC-05-16-2007_revision.pdf)). The NASS report represents another useful data source but also
14 would require extrapolation across reported crop acreage to represent a complete sample
15 of application rates.

16
17 The UFTRS system was not designed to track the source of inorganic nutrients applied to
18 agricultural land on the geographic scale needed for watershed modeling. The system
19 only tracks sales of synthetic fertilizers and not manure or biosolids applied to farmland.
20 In addition, geographical data associated with each may or may not be near the actual
21 point of application. However, given either regulatory or legislative changes (data
22 reporting is mandated through each state’s Fertilizer law) it could be possible to refine the
23 current system used by each State Department of Agriculture to generate more precise
24 data for improved modeling of watershed-scale nutrient mass balances. Those changes
25 would help target interventions and extension programs to improve nutrient management
26 and reduce nutrient losses. The lack of potential funding and the necessity to coordinate
27 all the states involved limit the practicality of such an approach.

28
29 The State Departments of Agriculture have already made recommendations to improve
30 the reporting system. These include:

- 31
32 1. an assessment to determine the needs for fertilizer usage data, the accuracy of the
33 current data collection methods, and whether methods require revision to meet
34 highest priority needs,
35
36 2. improvements in the database format and web-based access,
37
38 3. the identification of funding sources to support development of a more accurate,
39 accessible, and comprehensive database system, and
40
41 4. education and outreach to improve precision of reported fertilizer tonnage
42 including a clear distinction between nutrients used in crop, livestock, and non-
43 agricultural operations.
44

45 In addition, the information could be refined to reflect site-specific data layers, although
46 that would require development of a geospatial framework (and legal authority) to

1 encourage reporting at the retail level where it is possible to collect geographic
2 information.

3
4 The Chesapeake Bay watershed provides a good example of the fertilizer data dilemma.
5 While the fertilizer tonnage that is currently being utilized to calibrate the Chesapeake
6 Bay Program Watershed Model is relatively accurate, the county-specific tonnage may
7 have an accuracy of only ± 20 to 50%. For example, in a recent year, 17% of the reported
8 tonnage had been reported without an identified use and there are indications some
9 tonnage may have been reported more than once through the distribution chain
10 (Chesapeake Bay STAC report, Oct. 2007). It is also possible that fertilizer reported for
11 crop agriculture may actually have been used for lawn and turf, forestry, or other non-
12 agriculture applications.

13
14 Nitrogen fertilizer application data on a specific crop by crop basis that can be associated
15 with crop yields and location are essential for assessing both use patterns and efficiency.
16 The USDA-NASS maintains a database on N fertilizer rates applied to the major crops
17 (corn, wheat, cotton, soybeans, and occasionally other crops) based on farmer surveys
18 conducted every other year. These data represent another source of information
19 (“Protocols for Farming Reporting” Mark R. Miller, USDA, NASS).

20
21 Data derived from NASS farmer surveys include six categories: field crops, fruits and
22 vegetables, nurseries/floriculture, livestock use and post-harvest application. For each
23 group, NASS collects fertilizer, pesticide, and pest management data every year on a
24 stratified random sample of farmers at the field level. One field represents an entire farm
25 for each sample in the field crops survey. Fruit and vegetable information are collected
26 for the entire farm. If the field chosen for sampling has had manure applied in
27 conjunction with inorganic fertilizer, only the inorganic portion will be reported because
28 the survey does not ask about manure. Core crops are surveyed every other year on an
29 even/odd basis for different crops and surveyed states are selected to cover at least 80%
30 of planted acres. NASS is not currently scheduled to resume coverage of corn and other
31 commodity crops until 2010, which is a five year gap. NASS will try to resume its
32 coverage of corn management survey in 2008. This is a critical data gap and it is a
33 problem given the large changes in corn price and production area during this period.
34 Those data have to be available if there is to be progress in assessing fertilizer use and
35 efficiency for major crops in the U.S.

36
37 Based on these data, USDA has recently released an updated report on fertilizer use that
38 provides data on fertilizer consumption and type of fertilizer used from 1960-2006
39 (Figure 3-3 and Table 3-3. (US FERTILIZER USE AND PRICE; Released Friday,
40 October 5, 2007). Share of crop area receiving fertilizer and fertilizer use per receiving
41 acre, by nutrient, are presented for the major producing states for corn, cotton, soybeans,
42 and wheat. Additional data include fertilizer farm prices and indices of wholesale
43 fertilizer price. See <http://www.ers.usda.gov/Data/FertilizerUse/>

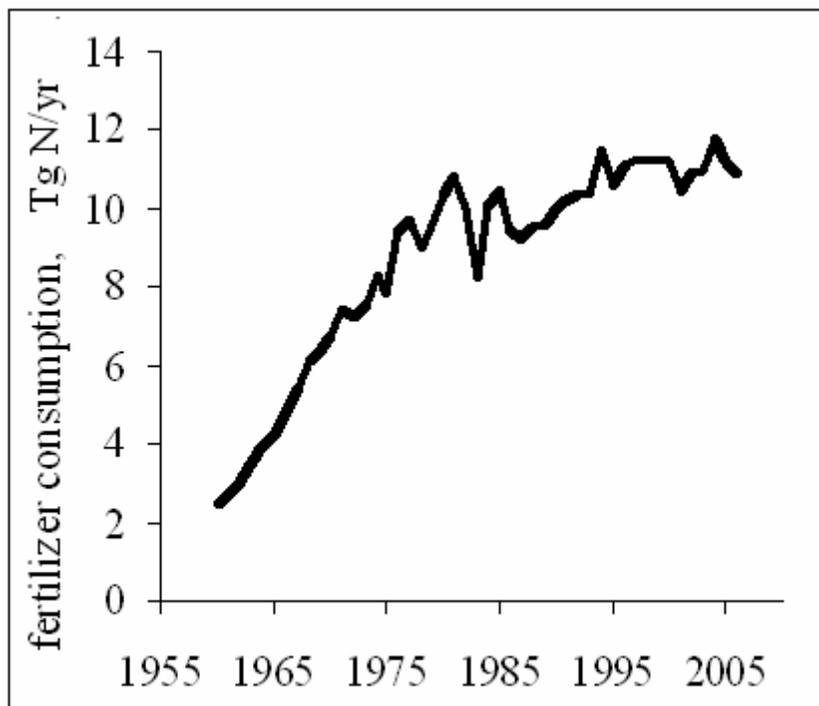


Figure 3-3 Fertilizer consumption in the US 1960 to 2006

<u>Synthetic Nitrogen Fertilizers</u>	Tg /yr	<u>% of total</u>
Other	0.21	2
Urea	2.21	20
Nitrogen Solutions	2.55	23
Anhydrous Ammonia	2.88	26
DAP, MAP, and NPK blends	2.28	21
Ammonium Sulfate, Ammonium Thiosulfate, and Aqua Ammonia, and Ammonium Nitrate	0.76	7
Total	10.89	100

Table 3-3. Sources and amount of nitrogen fertilizers used in the U.S. in 2002. Data from Terry et al. (2006).

Finding F3-2: Crop agriculture receives 63% of USA 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 USA Nr losses from terrestrial systems to air and aquatic ecosystems yet current monitoring of fertilizer use statistics by federal agencies is

1 inadequate to accurately track trends in quantities of N applied to major crops and the
2 geospatial pattern by major watersheds.

3
4 **Recommendation R3-2:** Improve detail and regularity of data acquisition for fertilizer
5 use by major crop (and for urban residential and recreational turf) and county (or
6 watershed) to better inform decision-making about policies and mitigation options for
7 nitrogen in these systems, and to allow monitoring of impact from implemented policies
8 and mitigation efforts.

9
10 Nitrogen fertilizer efficiency (NUE) is critical because higher use efficiency leaves less N
11 remaining to create potential environmental problems. Here and throughout this report we
12 define NUE as the grain yield per unit of applied N. All else equal, when higher NUE is
13 achieved without yield reduction, the crop takes up more of the applied N and
14 incorporates it into its biomass, which leaves less of the applied N at risk for losses via
15 leaching, volatilization, or denitrification. Fixen (2005) reports that there is substantial
16 opportunity for increasing NUE through development and adoption of more sophisticated
17 nutrient management decision aids.

18
19 A recent review of N-use efficiency for cereals from research field studies around the
20 world, mostly conducted on “small-plot” experiments at research stations, reported mean
21 single year fertilizer N recovery efficiencies for maize, wheat and rice of 65%, 57% and
22 46%, respectively (Ladha et al., 2005). However, crop fertilizer N recoveries based on
23 actual measurements in production-scale fields are seldom greater than 50% and often
24 less than 33%. For example, a review of N fertilizer recovery in different cropping
25 systems, (Cassman et al., 2002) estimated average recoveries of 37% for maize in the
26 north central U.S.

27
28 However, there are relatively few data that provide direct measurement of N fertilizer
29 recoveries by our major field crops under production-scale conditions and reducing the
30 uncertainty in estimates of NUE is fundamental for prioritization of research and
31 education investments, both in the public and private sectors. While management can
32 substantially improve NUE on average, in any given year weather will always be an
33 uncontrolled factor that can significantly influence system efficiency through effects on
34 crop growth vigor and ability to acquire applied nutrients, and on losses of nutrients due
35 to runoff, denitrification, and leaching that can occur in periods of excessive rainfall.

36
37 Although total N fertilizer use in the U.S. has leveled off in the past two decades (Figure
38 3-4, yields of all major crops have continued to increase. Because crop yields are closely
39 related to N uptake (Cassman et al., 2002), these trends imply a steady increase in NUE
40 and reduced N losses because more of the applied N is held in crop biomass and
41 harvested grain. Greater NUE has resulted from two factors: (1) a steady improvement in
42 the stress tolerance of corn hybrids (Duvick and Cassman, 1999) that increase crop
43 growth rates and allows sowing at higher plant densities, which together accelerate the
44 establishment of a vigorous root system to intercept and acquire available N in the soil
45 profile, and (2) the development and adoption of technologies that improve the
46 congruence between crop N demand and the N supply for indigenous soil resources and

1 applied N. Examples of such technologies include soil testing for residual nitrate and
2 adjusting N fertilizer rates accordingly, split N fertilizer applications, fertigation (the
3 application of nutrients through irrigation systems), site-specific management, and new
4 fertilizer formulations (e.g. controlled release, nitrification inhibitors). For maize, which
5 receives the largest share of total N fertilizer in the US, there has been a 38% increase in
6 N fertilizer use efficiency since 1980 (Figure 3-4). Similar improvements have been
7 documented for rice production in Japan and for overall crop production in Canada.

8
9 Despite these steady improvements, current levels of N fertilizer uptake efficiency appear
10 to be relatively low (Cassman et al., 2002), although data from production-scale studies
11 are few, and yet most farmers do not use best management practices (BMPs) with regard
12 to nitrogen fertilizer management. This situation suggests substantial potential for
13 improvement in NUE and an associated reduction in Nr losses from crop agriculture.

14
15 **Finding F3-3:** USDA statistics document that a large proportion of farmers do not follow
16 current BMPs for N, which indicates a tremendous opportunity to get the current non-
17 adopters of BMPs to adopt

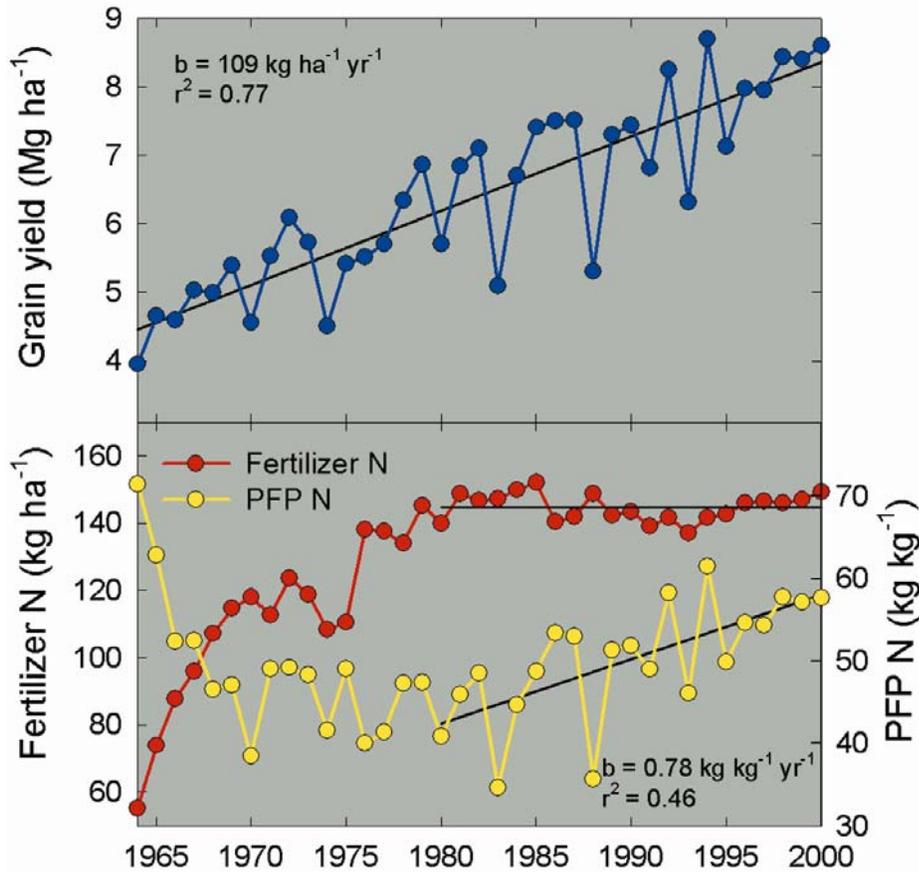
18
19 **Recommendation R3-3:** Several options (non-exclusive) would improve adoption rates:

- 20
21 1) expand the watershed-based “natural resource district” NRD system used in
22 Nebraska more widely to address Nr load from agriculture,
23
24 2) Subsidizing smart fertilizers and enhanced efficiency products,
25
26 3) Nutrient management plan for all farms,
27
28 4) increased efforts in extension to increase adoption rates of improved technologies
29 and farmer participatory research like those sponsored by the Iowa Soybean
30 Grower’s Associations with an explicit focus on improving NUE (website
31 citation)
32

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

40
41 A systematic effort needs to be made to update those data. The concept of NUE should be
42 emphasized as a way to address the need to balance economic *and* environmental goals.
43 In fact, the development and adoption of technologies that improve nitrogen fertilizer
44 efficiency can contribute to more profitable cropping systems through a reduction in
45 fertilizer costs. For example, average US nitrogen fertilizer efficiency required 0.45 kg
46 of applied nitrogen to produce 19.1 kg of grain yield in 1980 whereas that same amount

1 of nitrogen produced 26.5 kg of grain in 2000 (units converted from Figure 3-4). This
 2 gain in efficiency means that



3
 4
 5 **Figure 3-4.** Trends in US corn yields, N fertilizer rates applied to corn, and N fertilizer
 6 efficiency (kg grain produced per kg N fertilizer applied). (Cassman et al., 2002)

7
 8 it is possible to achieve the 2004 US average corn yield of about 150 bu/ac with 144 lbs
 9 of applied N fertilizer based on the nitrogen fertilizer efficiency achieved in 2000, versus
 10 about 200 lbs of nitrogen fertilizer at the 1980 efficiency level. At a cost of \$0.40 per
 11 pound of applied N, this reduction in nitrogen fertilizer input requirements represents a
 12 saving of about \$22/ac.

13
 14 Nitrogen’s strong positive impacts on yields in crops (e.g. corn) creates a strong
 15 economic incentive for its use. Nitrogen costs have increased dramatically since
 16 Hurricane Katrina, while corn prices have also increased dramatically with the increase in
 17 corn-based ethanol plants. However, the critical factor is the corn-to-fertilizer price ratio.
 18 If corn brings \$4.00 per bushel (25.5 kg) and nitrogen costs \$0.40 a pound (0.45 kg), this
 19 is a 10 to 1 price ratio – not different from the \$2.00 corn and \$0.20 nitrogen ratio that
 20 was typical from 2000-2005. There are also other critical factors in the farmer’s nitrogen
 21 application decisions such as yield at the margin and weather. In the Corn Belt, one or
 22 two years in five may provide extremely favorable weather for corn production. A

1 producer may view applying some extra nitrogen, hoping for good weather, as a
2 reasonable economic gamble. If the yield is more than half a bushel (12.7 kg) of corn per
3 pound (0.45 kg) of nitrogen at the margin or if there is more than one extremely good
4 year in five, the farmer comes out ahead.

5
6 Realistically, few farmers calculate their marginal returns from additional nitrogen in
7 good years versus average, but the high corn-to-fertilizer price ratio encourages some
8 farmers to plan for a good year and consider a larger nitrogen application than might
9 otherwise be appropriate for the nitrogen utilization in the four years of lower yield. This
10 presents a real dilemma if the policy goal is to reduce nitrogen losses, especially in the
11 four years of average or lower yields. Meeting this challenge will require approaches
12 such as the development of real-time, in-season, decision-making tools that allow crop
13 producers to use nitrogen fertilizer rates for average yields at planting and during early
14 vegetative growth, and a final top-dressing as required to meet any additional nitrogen
15 demand above this amount due to favorable climate and soil conditions that support
16 higher than average yields (Cassman et al, 2002; Cassman, 1999). Robust crop simulation
17 models using real-time climate data at a relatively localized geographic scale will be
18 required to develop such tools.

19
20 Another option is to develop new, alternative crop production systems that require less N
21 fertilizer. Such systems may employ legume cover crops, more diverse crop rotations,
22 and tighter integration between crop and livestock production to achieve greater reliance
23 on nitrogen inputs from legume N fixation and recycling of N in manure and compost .
24 At issue, however, is whether such systems actually reduce Nr losses to the environment
25 because the same loss mechanisms and pathways operate on nitrogen from both
26 commercial fertilizer and organic sources. Also at issue is the indirect land use change
27 impact from widespread adoption of these more diverse cropping systems because they
28 have reduced crop yields per unit land area compared to more simplified crop rotations
29 such as corn-soybeans that receive N fertilizer. Lower yields would require more land in
30 production to meet food demand. Therefore, a key issue is whether the tradeoff in
31 reduced N fertilizer inputs to more diverse crop rotations with organic N inputs would
32 actually result in less Nr losses compared to conventional cropping systems that require
33 less land to produce the same amount of crop output.

34
35 **Finding F3-4:** Nr inputs to crop systems are critical to sustain crop productivity and soil
36 quality. Moreover, given limited land and water resources, global population growth and
37 rapid economic development in the world's most populous countries, the challenge is to
38 accelerate increases in crop yields on existing farm land while also achieving a
39 substantial increase in N fertilizer uptake efficiency. This process is called "ecological
40 intensification" because it recognizes the need to meet future food, feed, and fiber
41 demand of a growing human population while also protecting environmental quality and
42 ecosystem services for future generations (Cassman, 1999). More diverse cropping
43 systems with reduced N fertilizer input may also provide an option if the tradeoff
44 between lower yields per unit land area and time is more than offset by the reduction Nr
45 losses per unit of crop production to avoid expansion of crop production area to meet
46 demand.

1
2 **Recommendations R3-4:**
3

- 4 1) Data on NUE based on direct measurements from production-scale fields are
5 required for the major crops to identify which cropping systems and regions are of
6 greatest concern with regard to mitigation of Nr load to better focus research
7 investments, policy development, and prioritization of risk mitigation strategies.
8
9 2) Develop policies and incentives that promote the use of “smart” (controlled
10 release) nitrogen fertilizers that have potential for substantial reduction in Nr
11 losses without a negative impact on crop productivity.
12
13 3) Research is needed with an explicit focus on the challenge of *both* accelerating
14 the rate of gain in crop yields on existing farm land while substantially increasing
15 N fertilizer uptake efficiency, and also on quantifying whether widespread
16 adoption of lower-yielding more diverse cropping systems with lower N fertilizer
17 input requirements can reduce regional Nr load when the impact of indirect land
18 use change is considered.
19
20 4) EPA should work closely with the U.S. Department of Agriculture (USDA),
21 Department of Energy (DOE), and the National Science Foundation (NSF) to help
22 identify research and education priorities for prevention and mitigation of Nr
23 applied to agricultural systems.
24

25 *3.2.3.2. Biological fixation in cultivated croplands.*
26

27 Reactive N is also introduced to the landscape in significant quantities via biological N
28 fixation (BNF) in cultivated crop lands. Management of biologically fixed nitrogen,
29 insofar as it is possible, is proportionally as critical a task as the management of synthetic
30 N because Nr from BNF is prone to the same loss pathways as Nr from commercial
31 fertilizers. To quantify BNF due to human cultivation of crops, we calculated the annual
32 agricultural fixation for 2002 using crop areas and yields reported by the Census of
33 Agriculture (2002). We multiplied the area planted in leguminous crop species by the
34 rate of N fixation specific to each crop type, assigning rates based on a literature review,
35 as summarized in Table 3-4 below and shown relative to other inputs in Table 3-1.
36 Annual nitrogen inputs to cropping system from BNF by legume crops was 7.7 Tg N/yr
37 in 2002, accounting for ~15% of the overall Nr inputs to the terrestrial landscape from all
38 sources, and 20% of the agricultural sources (Table 3-1). Soybean and alfalfa
39 contributions are the most important agricultural legumes in terms of nitrogen input and
40 contribute 69% of total BNF inputs in US agriculture.
41

Nr fixation in cultivated croplands

	production area, Mha	rate, kg/ha/yr	Tg N/yr	% of total
soybeans	29.30	111	3.25	42
alfalfa	9.16	224	2.05	27
other leguminous				
hay	15.37	117	1.80	23
western pasture	161.82	1	0.16	2
eastern pasture	21.99	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.

Table 3-4. Estimates of nitrogen input from biological nitrogen fixation (from major legume crops, hay, and pasture)

3.2.3.3. 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 nitrogen efficiently. For example, denitrification can range from 0-70% of applied N. 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 nitrogen 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 nitrogen fertilizer (uncertainty range of 0.3-3.0%) is lost from direct emissions of N₂O 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 N₂O emission is applied to other nitrogen 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, and a number of recent studies confirm that N₂O 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 N₂O losses can vary widely even within the same field and from year to year due to normal variation in climate and crop management.

Additional indirect N₂O emissions result from denitrification of volatilized ammonia-N deposited elsewhere, or nitrate lost to leaching and runoff as the Nr cascades through other ecosystems after leaving the field to which it was applied. Here the IPCC

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1 assessment protocol assumes that volatilization losses represent 10% of applied nitrogen,
2 and that N₂O emissions for these losses are 1% of this amount; leaching losses are
3 assumed to be 30% of applied nitrogen and N₂O emissions are 0.75% of that amount
4 (IPCC, 2007). Therefore, the IPCC default value for total direct and indirect N₂O
5 emissions represents about 1.4% of the applied nitrogen from fertilizer. By the same
6 calculations, 1.4% of the nitrogen in applied organic matter, either as manure or compost,
7 or in recycled crop residues, is also assumed to be emitted as N₂O.

8
9 Others have estimated higher average N₂O losses of 3-5% of applied nitrogen fertilizer
10 based on global estimates of N₂O emissions from recycling of Nr (Crutzen et al., 2007),
11 as opposed to the field-based estimates that form the basis of IPCC estimates. Because
12 N₂O is such a potent greenhouse gas, and given the more than 2-fold difference in
13 estimates of N₂O losses, there is a critical need to improve understanding and prediction
14 of N₂O losses from agricultural systems. N₂O emissions in the US are estimated to be
15 0.78 Tg N/yr (Table 3-5) (USEPA, 2005).

16
17 Biogenic NO_x emissions from croplands are on the order of 0.5% of fertilizer input—
18 much more than this in sandy soils and less as clay content increases (Aneja et al. 1996;
19 Sullivan et al. 1996; Veldkamp and Keller. 1997; Civerolo and Dickerson, 1998).
20 However, NO_x emissions by agricultural burning are relatively unimportant. Ammonia
21 volatilization of N from applied fertilizer can be the dominant pathway of N loss in rice
22 soils and can account for 0->50% of the applied N depending on water management, soil
23 properties and method of application (citations within Peoples et al. 1995). Ammonia
24 volatilization can be of the same range in upland cropping systems, with largest losses
25 occurring typically on alkaline soils (Peoples et al. 1995). The IPCC (2007) uses a value
26 of 10% of synthetic fertilizer N application and 20% of manure N as estimates of average
27 ammonia volatilization.

28
29 Taken together, N losses from all forms of direct gaseous emissions forms from crop
30 production systems can represent a substantial portion of applied N fertilizer when soil
31 conditions favor such emissions and there is a lack of synchrony between the amount of
32 N applied and the immediate crop demand (Goulding, K., 2004). Therefore, achieving
33 greater congruence between crop demand and the N supply from fertilizer is a key
34 management tactic to reduce N losses from all sources. Success in reducing N losses and
35 emissions from agriculture will depend on increased efforts in research and extension to
36 close gaps in our understanding of N cycling and management in crop production,
37 especially as systems further intensify to meet rapidly expanding demand for food, feed,
38 fiber, and biofuel.

	<u>Tg N/yr</u>	<u>%</u>
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

Table 3-5. N₂O emissions in the USA, 2002

Finding F3-5: 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 GHG 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 R3-5: We recommend 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.

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

Our current transportation system requires enormous amounts of liquid motor fuels at a time when petroleum use exceeds petroleum discovery. Hence, the price of petroleum has increased more than five-fold in the past 10 years. Most of the world’s petroleum reserves are located in politically unstable countries, this further boost prices due to supply uncertainty. This situation provides strong motivation for investment in biofuels made from crops, and a number of countries have enacted favorable policies and incentives to foster a rapid expansion of biofuel production capacity. In the USA, ethanol production from corn has doubled to 30 billion liters/yr since 2005, and biorefineries to produce an additional 20 billion liters/yr are currently under construction. Brazil is rapidly expanding its production of sugarcane ethanol, Europe and Canada are expanding biodiesel production from canola oil, and Indonesia and Malaysia expect to greatly increase biodiesel production from palm oil.

At current petroleum prices, the highest value use of corn is as feedstock for biofuel rather than for human food or livestock feed (CAST, 2006). As a result, the amount of corn used for ethanol is rising rapidly and about 30% of USA corn production will be used for ethanol in 2008, which represents about 10% of global corn supply. Due to this increased demand, corn prices have risen about three-fold since 2005, which sends powerful signals to corn producers to increase production. Farmers have responded by

1 increasing corn acreage by an additional 10-12 million acres, and they may be motivated
2 to increase N fertilizer rates to boost yields. However, N fertilizer prices have also risen
3 so the net impact of expanded biofuel production on actual N rates used by crop
4 producers is uncertain. Production of large amounts of distillers grains co-product is also
5 changing the way in which livestock feed rations are formulated, which in turn could
6 have a large influence on the cycling of N in cattle manure.

7
8 **Finding F3-6:** Rapid expansion of biofuel production and high commodity prices are
9 changing the cost-benefit ratio of N fertilizer use in crop production and also changing
10 the nutrient profile of livestock diets with consequences for effective management of Nr.

11
12 **Recommendation R3-6:** Critical need to understand and predict these changes in terms
13 of maximizing the N efficiency of both crop and livestock production systems, and to
14 develop strategies for avoiding increased Nr load in the environment as a result of current
15 and future expansion of biofuel production from corn and other “second generation”
16 biofuel feedstock crops.

17 18 **3.2.4. Nr inputs and losses from animal agriculture**

19
20 In the US, domestic animals produce 6.0 Tg N/yr in manure and are the largest source of
21 atmospheric NH₃-N (1.6 Tg N/yr) (Table 3-1). Livestock also contribute to N₂O-N
22 emissions, though in much smaller proportions (~4% of total US N₂O-N emissions).

23 24 *3.2.4.1 Trends in Animal Agriculture*

25
26 While animal production has been increasing since World War II, this report will
27 emphasize the period from 1970 to 2006. The production of chicken broilers increased
28 by more than four fold from 1970 to 2006 (Figure 3-5) and milk production increased by
29 nearly 60% in this time period (Figure 3-5). Turkey production doubled and pork
30 production increased about 25%, while meat from cattle (beef and dairy) remained
31 constant (Figure 3-5).

32
33 Another trend in animal production has been for fewer animals to produce more animal
34 products. For example, the 60% greater amount of milk produced in 2006 compared to
35 1970 required 25% fewer cows (Figure 3-6). Animal inventories declined by 10% for
36 beef brood cows from 36 million head in 1970 to 33 million head in 2006, and the
37 inventory of breeder pigs and market hogs declined 8% from 673 million head to 625
38 million head in the same period, even with similar or greater annual meat production.
39 This trend resulted from greater growth rates of animals producing more meat in a shorter
40 amount of time. In 1970, broilers were slaughtered after 80 days on feed at 1.7 kg live
41 weight, but by 2006 the average weight was 2.5 kg after only 44 days on feed (NASS-
42 USDA, 2007).

43
44 Another trend in animal agriculture has been the increased size and smaller number of
45 animal operations, which results from the mechanization of agricultural practices and
46 increased specialization. There were only 7% as many swine operations and 11% as

1 many dairy operations in 2006 as there were in 1970 (Figure 3-7). There were half as
2 many beef operations in 2006 as in 1970, but beef operations also expanded in size while
3 smaller producers held jobs off the farm.

4
5 All of these trends show an increase in labor efficiency to produce a similar or greater
6 amount of animal products. Also, because animal production is more concentrated on
7 fewer farms with greater specialization, fewer crops are produced on those farms. As a
8 result, it is increasingly common to have more manure nutrients produced on a livestock
9 farm than can be used efficiently as fertilizer for crops on that farm. Therefore, unless the
10 manure is applied over a larger crop area, the resulting over-application of manure on the
11 livestock farm can reduce the subsequent efficiency of its utilization and result in greater
12 nutrient losses.

13 14 *3.2.4.2. Impact of Livestock Production Trends on Nitrogen Use Efficiency*

15
16 The trends have both positive and negative environmental impacts. One of the significant
17 positive impacts is that with smaller animal inventories producing greater quantities of
18 animal products, there is an improved efficiency of nutrient utilization per product
19 produced. This effect is partly the result of effectively reducing maintenance
20 requirements during production. The requirements for feeding animals can be divided
21 into two components: maintenance and production. The maintenance component is that
22 feed which is used to keep the animal alive and healthy so that production is possible.
23 The production component includes feed that is converted to animal protein and waste
24 due to the inefficiencies of these conversions. The maintenance component depends
25 upon the number of animals, each animal's mass, and the time the animal is on feed.
26 Thus, the maintenance requirement is diluted by faster growth rates and greater body
27 weight at slaughter. The increases in production rates over time have led to greater
28 efficiencies in N and P utilization for animal production, and lower amounts of nutrients
29 excreted per unit of animal protein produced.

30
31 Public concerns about the potential environmental and health effect of air emissions from
32 concentrated animal feeding operations (CAFOs) expand the impacts of food production
33 beyond those associated with traditional agricultural practices. Increased emissions of
34 nitrogen compounds from animal agriculture into the atmosphere may lead to increased
35 odor, and interact in atmospheric reactions (e.g. gas-to-particle conversion) (Baek et al.
36 2004a; Baek and Aneja 2004b). These are then transported by wind and returned to the
37 surface by wet and dry deposition processes, which may have adverse effects on human
38 health and the environment (McMurry et. al, 2004; Aneja et. al, 2006, 2008a, b, c;
39 Galloway et. al, 2008).

40
41 Adverse effects include aerosol formation, soil acidification, eutrophication, loss of
42 biodiversity, and the neutralization of acids produced by sulfur and nitrogen oxides.
43 Aerosol formation occurs when ammonia reacts with other acidic compounds in the air.
44 When ammonia is deposited to soils, it can be oxidized by soil microbes to produce the
45 nitrate ion. Thus soil acidification occurs when ammonia deposits on soils with low
46 buffering capacity, which can cause growth limitations to sensitive plant species.

1 Deposition of ammonia also causes eutrophication (i.e. an over-abundance of nutrients),
2 which can promote harmful algal growth leading to the decline of aquatic species. In
3 fact, volatilized ammonia can travel hundreds of miles from its source affecting local and
4 regional biodiversity far from its origin (Aneja et al. 2008b; James, 2008).

5
6 The potential for reduced environmental impact from Nr in livestock systems depends on
7 the proportion of the total intake attributable to maintenance costs. The commonly used
8 tables for diet formulation published periodically by the National Research Council
9 (NRC) for various animal commodities can be used to track diet formulation practices
10 and assumptions regarding maintenance and production requirements. About one third of
11 the energy intake recommended for growing broilers was assumed to be needed for
12 maintenance (NRC, 1994) but protein requirements were not divided between
13 maintenance and production. For example, a dairy cow producing 40 kg milk per annum
14 would divert about 25% of its energy and 12% of its protein to maintenance (NRC,
15 1989).

16
17 In terms of nutritional efficiency of a herd or flock, maintenance of a productive phase
18 (e.g. growth, lactation) also requires maintenance of a reproductive phase of the animals
19 life cycle. In other words, the actual nutritional maintenance cost of a herd or flock is
20 greater than it is for productive individuals only. For example, milk production requires
21 non-lactating cows and heifers in the herd which do not produce milk but which consume
22 nutrients. These additional maintenance costs are lower for broiler flocks than for cattle.

23
24 **Finding F3-7:** There are no nationwide monitoring networks in the US to quantify
25 agricultural emissions of greenhouse gases, NO, N₂O, reduced sulfur compounds, VOCs,
26 and NH₃. In contrast there is a large network in place to assess the changes in the
27 chemical climate of the US associated with fossil fuel energy production, ie the National
28 Atmospheric Deposition Program/National Trends Network (NADP/NTN), which has
29 been monitoring the wet deposition of sulfate (SO₄²⁻), nitrate (NO₃⁻), and ammonium
30 (NH₄⁺) since 1978.

31
32 **Recommendation R3-7:** The status and trends of gases and particulate matter emitted
33 from agricultural emissions, eg ammonia/ammonium may be monitored nationwide by a
34 network of monitoring stations.

35 36 37 *3.2.4.3. Changes in Feeding Practices*

38
39 From 1970 to 2006, several feeding practices were changed for diets fed to livestock. In
40 1989 and 1996, the National Research Council introduced the idea of dividing the form
41 of protein fed ruminants into that which is degraded by rumen microorganisms and that
42 which passes through the rumen to be digested directly in the stomach and small
43 intestine. Feeding ruminants with attention to rumen degraded and rumen undegraded
44 protein decreases the amount of protein fed by 10 to 15% for a given protein requirement.
45 For poultry and swine, manufactured amino acids were added to diets, decreasing the
46 need for protein by 30%. Today, two amino acids (lysine and methionine), coated in a

1 way to prevent degradation in the rumen, are sometimes added to dairy cattle diets
2 thereby decreasing protein intake by another 15% (NRC, 2001). Phytase added to swine
3 and poultry diets in the past decade has decreased phosphorus feeding by 20 to 50% with
4 some of the decrease attributed to simply better understanding phosphorus requirements
5

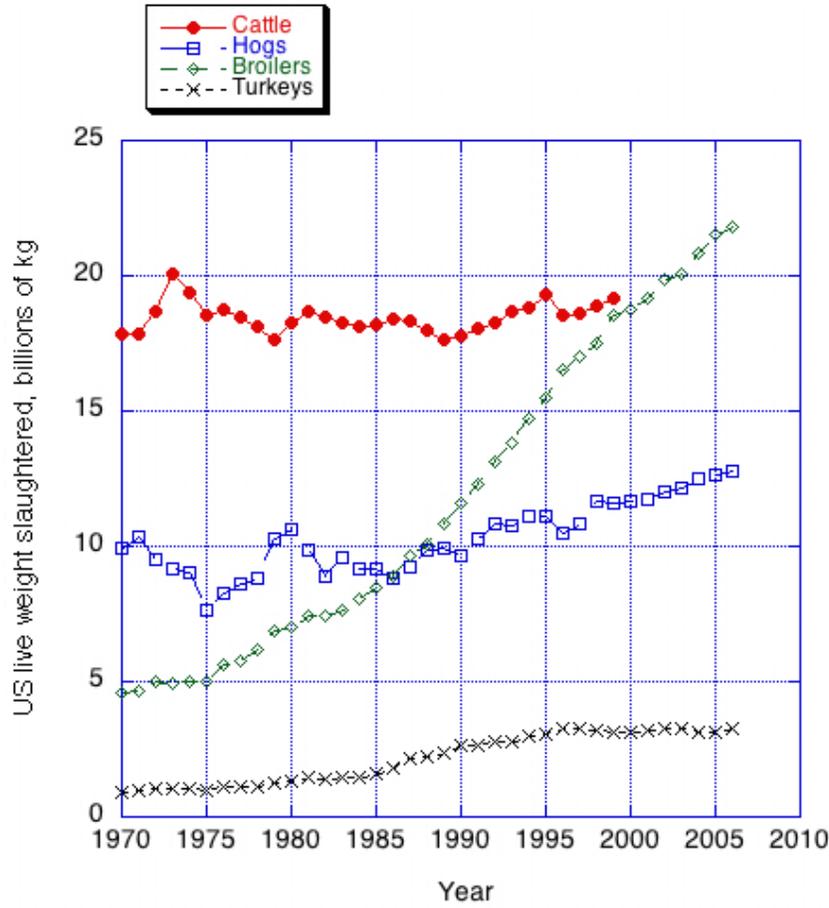
6 It is difficult to estimate the combined effects of changes in feeding practices, but for
7 calculations on changes in manure N, we assume improvements in both production rates
8 and ration formulation. In the case of beef cattle diet formulation, the changes in feeding
9 practices were determined by using NRC, 1976 compared to NRC, 1996. Surprisingly,
10 NRC 1996 recommended greater total crude protein compared to NRC 1976 despite
11 formulating for rumen degraded and undegraded protein and considering amino acid
12 content. Therefore, improved diet formulation did not decrease N intake for beef in this
13 time range but the effect of reduced maintenance did improve efficiency of N utilization.
14

15 *3.2.4.4. Nitrogen Excretion*

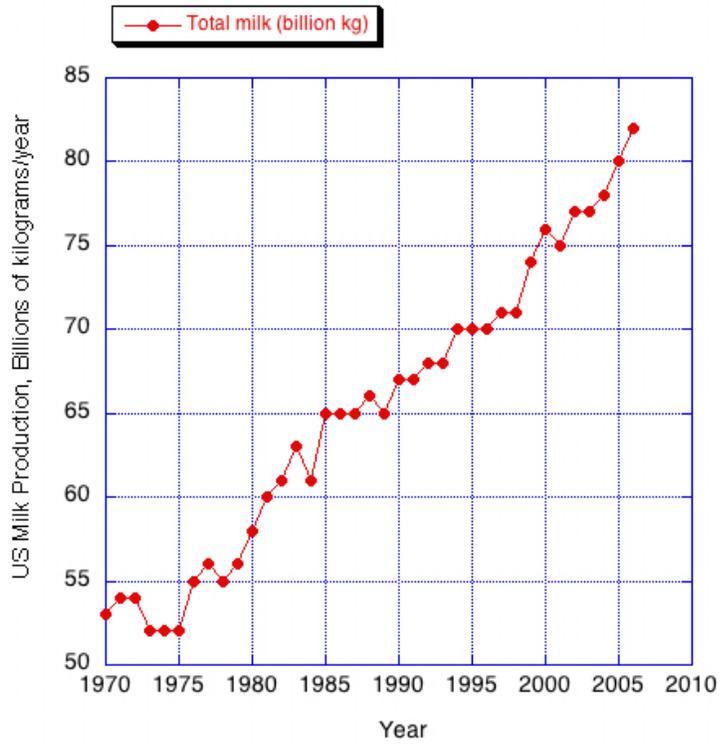
16
17 Nitrogen excretion as fraction of animal production decreased from 1970 to 2006 (Table
18 3-6). However, in cases where the total amount of animal production in the US increased
19 substantially (e.g. broilers), total N excretion increased. The decrease in N excretion per
20 unit of animal productivity was estimated by calculating the effects of changes in feeding
21 practices and reduction of maintenance as described previously and explained in detail in
22 the Appendix.
23

24 For broilers, data are available to more accurately estimate the effect of changes in
25 feeding and genetics on N excretion over time. However, these data do not represent the
26 time period of interest in this report. Havenstein et al. (1994) compared a 1957 strain of
27 broiler fed a 1957 diet to a 1991 strain fed a 1991 diet. Based on the reported N intake
28 and production data, there was a 51% reduction in N excreted between these diets (Kohn,
29 2004).
30

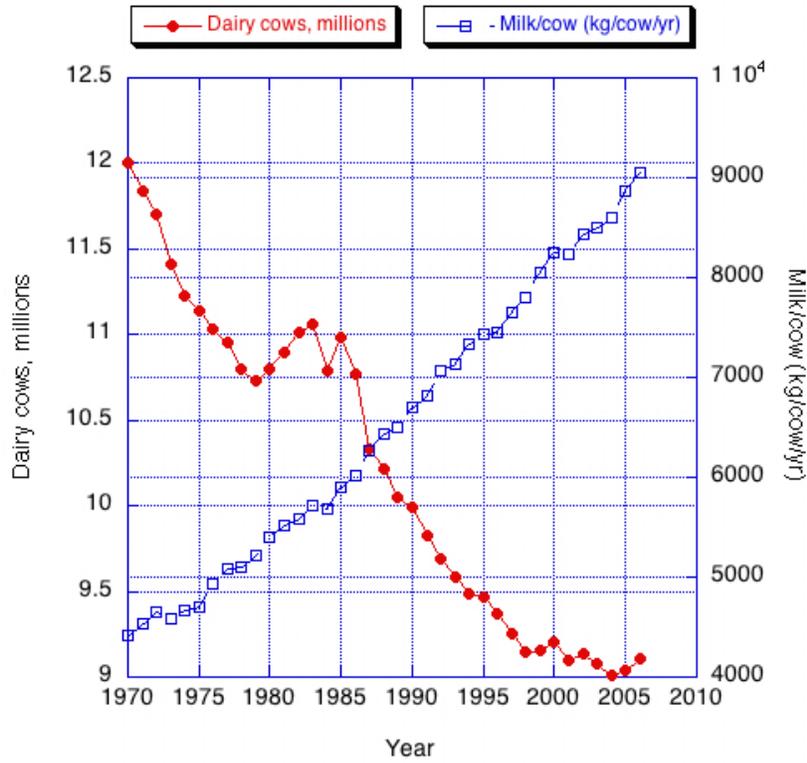
31 Similarly, Kohn (2004) compared N excreted by US dairy cows in 1944 and 2001. In
32 1944, the historically largest herd of dairy cattle in the US (25 million cows) produced an
33 average of 7 kg milk per cow per day (NASS-USDA, 2007). In 2001, nine million cows
34 produced an average of 27 kg milk per cow per day. Assuming the cows in 1944 and
35 2001 were fed according to popular feeding recommendations of the time, the N intakes
36 were 360 and 490 g/d per cow, and N excretion rates (N intake – N in milk) were 326 and
37 364 g/d per cow. Multiplying by the number of cows in the US, shows that total milk
38 production increased 40% from 52 billion kg to 73 billion kg, while N excretion
39 decreased 60% from 3.0 Tg N to 1.2 Tg N, respectively.
40
41
42
43



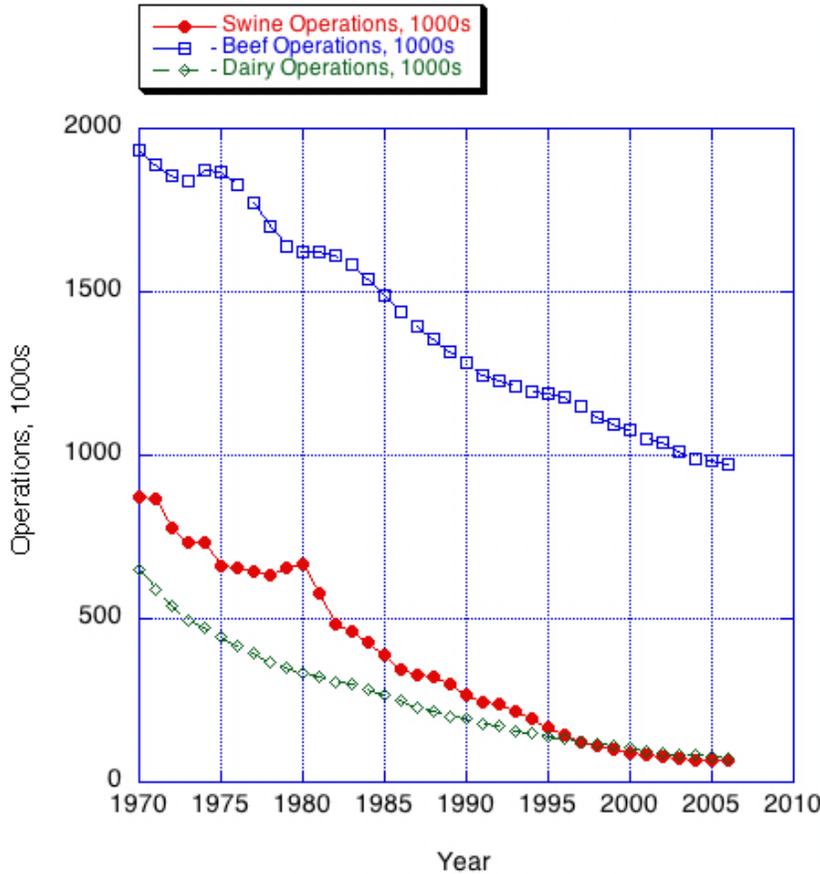
1
2 **Figure 3-5 Meat production from 1970 to 2006. Source: USDA-NASS, Census Reports.**
3



1
2 **Figure 3-6. Milk production from 1970 to 2006. Source: USDA-NASS, Census Reports.**



1
2 **Figure 3-7 US Inventory of mature dairy cows and milk production per cow from 1970 to 2006. Source:**
3 **USDA-NASS, Census Reports.**
4



1
 2 **Figure 3-8. Number of animal operations in the US from 1970 to 2006. Source: USDA-NASS, Census**
 3 **Reports.**
 4
 5

Commodity ¹	1970		2006	
	g/kg product	Total US	g/kg product	Total US
Milk	17	0.894	11	0.916
Pork, live weight	57	0.564	42	0.536
Broilers, live weight	56	0.256	46	1.005
Beef, live weight	123	1.205	111	1.320

¹ Does not include manure produced for reproduction of stock (e.g. growing dairy heifers, breeder pigs).

6
 7
 8 **Table 3-6 Manure N excreted per kg production (g/kg) and per total US (Tg /yr).**
 9
 10

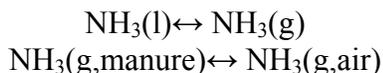
11 In Table 3-7, manure N was calculated for all U.S. animal agriculture using data on
 12 animal production from the 2002 Census of Agriculture (USDA 2002). For data on
 13 livestock production (cattle, calves, poultry, hogs, and pigs), manure was calculated by
 14 the methods of Moffit and Lander (1999), following the exact methods they had used to
 15 compute manure from the 1997 Census of Agriculture, but using the updated information
 16 from the 2002 Census of Agriculture. For data on production of manure from other
 17 animals (horses, goats, and sheep), we used coefficients for manure excretion as a
 18 function of average animal weights and animal inventory, taken from Battye et al. (1994).
 19

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

Table 3-7. Manure production from animal husbandry in the continental USA, Tg N per year 2002.

3.2.4.5. Volatilization of Animal Waste

Ammonia volatilization is highly variable and is influenced by the amount of total ammoniacal 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 ammonia volatilization. NH₃ increases linearly with TAN concentration. Higher temperatures increase NH₃ volatilization rates due to decreased solubility in turn affecting NH₃/NH₄⁺ equilibrium which follows Henry's law for dilute systems:



Ammonia-ammonium equilibrium [NH₄⁺(l) ↔ NH₃(l) + H⁺] is affected by temperature influencing the dissociation constant K_a [K_a = (NH₃)(H₃O⁺)/(NH₄⁺)] and pH. Levels of pH greater than 7.0 allow NH₃ to undergo volatilization. Otherwise, NH₃ is in the form of NH₄⁺ and therefore cannot be volatilized (Arogo et al., 2006; James, 2008).

The USEPA estimates annual manure N excreted in livestock production in the U.S. for the "Inventory of U.S. Greenhouse Gas Emissions and Sinks" (USEPA, 2007). For the year 2002 these estimates (Appendix 3, Table A-174; USEPA, 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 used as fertilizer 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 and was not recovered for further use. This N is susceptible to movement into the atmosphere and aquatic systems or incorporation into soil organic matter. By a combination of Best Management Practices (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).

Table 3-8 Fate of Livestock Manure Nitrogen (Tg N) (USEPA, 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

Total manure reported in Table 3-7 in the contiguous USA was estimated using USDA's method yield an estimate of 6.0 Tg N/yr; while EPA's greenhouse gas inventory method in Table 3-8 above yields a total for the USA 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 3-8 include Alaska and Hawaii whereas the values in Table 3-7 do not; though given the small relative amount of livestock production in those states that doesn't contribute substantially to the difference.

Finding F3-8: 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 R3-8: Policy, regulatory, and incentive framework needed to improve manure management to reduce Nr load and ammonia losses and also taking account of phosphorus load issues.

3.2.5. Nr inputs to residential and recreational turf systems

Turf grasses cover 12.6-16.2 million ha across the continental USA (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 (~75%) is in residential lawns. About 80% of all U.S. households have private lawns (Templeton et al. 1998) that average 0.08 ha in size (Vinlove and Torla, 1995). Another ~15 % of total turf grass area is in low maintenance parks and ~ 10% is in athletic fields and golf courses, which often receive higher levels of nitrogen application due to hard use conditions.

Supplemental nitrogen fertilization is often necessary to maintain healthy and aesthetically pleasing turf color, high shoot density and the ability to resist and recover

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1 from stress and damage. Nitrogen also may be derived from atmospheric deposition or
2 recycled decomposition of soil and grass clipping organic matter. Whether these inputs
3 are sufficient to maintain lawns of adequate quality depends on many factors including
4 age of the turf, uses, and expectations or goals of the homeowner or field manager. Also,
5 turf grasses are used to stabilize soil, often with an erosion prevention matrix such as
6 organic mats or with hydroseeding. Depending on circumstances, these turf uses may be
7 temporary until natural vegetation succeeds the turf, or may be low maintenance turfs that
8 are seldom fertilized such as highway medians and shoulders, grassy swales and buffers.

9
10 Turf grass is maintained under a variety of conditions. Approximately 50% of all turf
11 grass is not fertilized while the remainder is fertilized at varied intensities (Petrovic,
12 personal communication—June 5 2007). We have arrayed the different turf
13 managements into three groups according to the estimated amount of N-fertilizer applied
14 annually (Table 3-9), residential lawns maintained by homeowners (0.73 kg/100 m²),
15 residential lawns cared for by professional lawn care companies (2.92 (1.95-7.3) kg/100
16 m²), and athletic fields and golf courses (3.89 (2.64-6.64) kg/100 m²). The estimate of
17 total N-fertilizer used on turf grass in the USA is 1.11 Tg/year, or 9% of the total average
18 annual N-fertilizer used between 1999 and 2005. Depending on land use patterns, certain
19 areas of the country, particularly coastal areas where residential and urban properties
20 prevail, turf fertilizer can be an important, or even dominant source of nitrogen to surface
21 waters.

22
23 Turf fertilizer N is susceptible to loss to the atmosphere, and surface and ground water
24 when it is not properly managed. Research on lawns has shown that leaching of nitrate
25 can range between 0 and 50% of N applied. Nitrogen leaching losses can be greatly
26 decreased by irrigating lightly and frequently, using multiple and light applications of
27 fertilizers, fertilizing at the appropriate times, especially not too late in the growing
28 season, and using soil tests to ensure proper balance of non-nitrogen soil condition and
29 pH. In a soil column experiment with turf coverage, the percentage of nitrogen leached
30 (as percentage of nitrogen applied) varied from 8 to 14% using light irrigation and from 2
31 to 37% with heavy irrigation.

32
33 Applying fertilizer in appropriate amounts, avoiding periods when grass is dormant, and
34 not fertilizing too soon before irrigation or large rainfall events can all help ensure
35 leaching and runoff will be minimal without affecting turfgrass color and growth
36 (Mangiafico and Guillard, 2006).

37
38 Nitrogen runoff losses are poorly quantified but a range similar to leaching is probable
39 (Petrovic, personnel communication). The chemical form of fertilizer N does not impact
40 leaching/runoff unless it is applied in late autumn (Petrovic, 2004), although use of slow
41 release or organic fertilizers can help reduce runoff and leaching. Shuman (2002) notes
42 that runoff can be limited by applying minimum amounts of irrigation following fertilizer
43 application and avoiding application before intense rain or when soil is wet. Volatile
44 losses of ammonia can be significant when urea is applied. Measured denitrification
45 losses are usually small, but depend upon timing of N application relative to soil water
46 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/100m², 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 nitrogen 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 nitrogen supplements once mature. These practices can potentially reduce nitrogen 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 3-9. Estimate of Fertilizer N used on Turf grass in the USA in the year 2000, based on a total area of 12.6 million ha.

Type of Turf Fertilized N Used	Area (Million ha)	N Rate kg/ha*	Total Tg N
Nominal Fertilization	4.7	73.2	0.35
Professional Lawn Care	0.93	296 (195-488)	0.27
High Maintenance Areas (golf/sports)	1.26	390	0.49
Total	6.89	--	1.11

*10,000 m²/ha, used values of 0.73, 2.92 and 3.89 kg N/100 m² for nominal fertilization, professional lawn care, and high maintenance areas, respectively.

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3.3. Nr Transfer and Transformations in and between Environmental Systems (ES)

This chapter discusses the transfers and flows of reactive nitrogen (Nr) within and between environmental systems (ES) which include atmosphere, terrestrial, and aquatic environments. The first section (3.3.1) contains information on Nr deposition from the atmosphere to terrestrial and aquatic ES, presents estimates of input and recycling of Nr within terrestrial ES, and discusses movement of Nr from the terrestrial to the aquatic ES. The second section (3.3.2) presents an estimate of storage of Nr within the terrestrial ES. The total input of Nr into the continental USA and part of the transfers between environmental systems within the Nitrogen Cascade (Fig. 2-1) can be found in Table 3-1. Using these data we have constructed a “best guess” estimate of flows of Nr within and between ES in the Nitrogen Cascade in Section 3.3.3. Within the Nitrogen Cascade there are a number of places where the flow of Nr is constrained or regulated which we refer to as “control” points. These offer opportunities to remove excess Nr from the system. Important control points are discussed in Section 3.3.4. In the final section a list of critical information needs is presented.

3.3.1 Input and Transfers of Nr in the U.S.

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.

3.3.1.1 Nitrogen Deposition from the Atmosphere to the Earth’s Surface

Introduction. The magnitude and mechanisms of reactive N 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, NH₃ and their reaction products that are 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. 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 reactive N species. While most Nr deposited to the earth’s surface remains there, a fraction is re-emitted as NH₃, NO, or N₂O [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.

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1 In this section we review the state of the science concerning the total annual reactive N
2 deposition and trends in that deposition for to the contiguous 48 states.

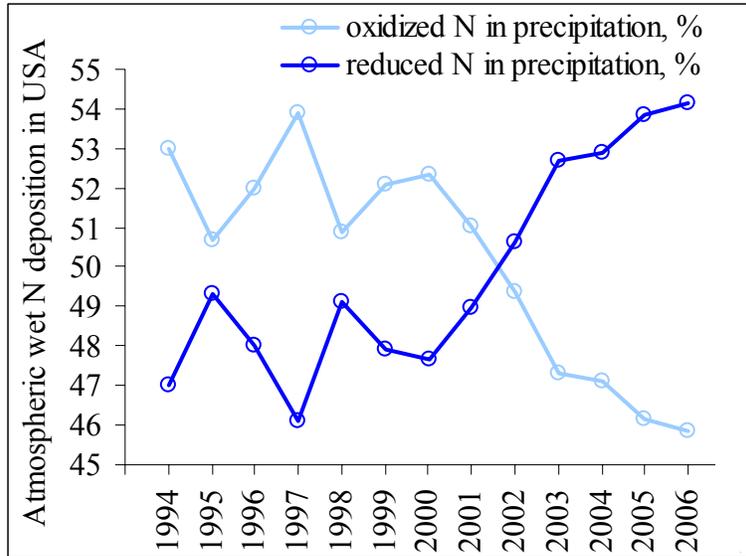
3
4 Deposition involves both oxidized and reduced N species. Of the oxidized forms of atmospheric
5 N, all the members of the NO_y family (NO, NO₂, NO₃, N₂O₅, HONO, HNO₃, NO₃⁻, PAN and
6 other organo-nitrates, RONO₂) can be transferred from the troposphere to the surface, and some
7 undergo bidirectional flux. Note that volatile amines are also detected as NO_y compounds
8 [*Kashihira et al.*, 1982; *Wyers et al.*, 1993]. Although a potent greenhouse gas, N₂O is only
9 emitted, not deposited and therefore will not be considered here. Of the reduced forms of
10 atmospheric nitrogen, NH₃ and NH₄⁺ play a major role. There is also evidence of deposition of
11 organic N such as amino acids and isoprene nitrates, and recent observations suggest that these
12 can account for as much as 10% (possibly 30%) of the US NO_x budget, especially in summer
13 [*Duce et al.*, 2008; *Horowitz et al.*, 2007; *Keene et al.*, 2002; *Sommariva*, 2008]. While this is a
14 worthy research topic, measurements are still limited and deposition of organic N compounds
15 will not be reviewed here. The wide array of relevant atmospheric compounds makes direct
16 measurement, and accurate load quantification challenging.

17
18 *Review of Nr Wet Deposition.* Substantial progress has been made in monitoring wet deposition,
19 as is summarized by the National Atmospheric Deposition Program/National Trends Network
20 (NADP), established in 1979, which monitors precipitation composition at over 250 sites in the
21 US and its territories (<http://nadp.sws.uiuc.edu>). Precipitation at each station is collected weekly
22 according to well established and uniform procedures from which it is sent to the Central
23 Analytical Laboratory for analysis of acidity, sulfate, nitrate, ammonium, chloride, as well as
24 base cations including calcium, magnesium, potassium and sodium. For greater temporal
25 resolution, the Atmospheric Integrated Research Monitoring Network AIRMoN, comprised of
26 seven sites, was formed in 1992 as part of the NADP program to study wet deposition
27 composition and trends using samples collected daily. The same species are measured as in
28 NADP. By interpolating among sites, NADP is able to estimate the wet deposition of
29 ammonium, and nitrate for the 48 contiguous states (Table 3-10. and Figure 3-9.).

30
31 **Table 3-10.** Annual deposition of reduced, oxidized, and total N to the 48 contiguous states, from
32 the NADP/NTN network <http://nadp.sws.uiuc.edu> .

33

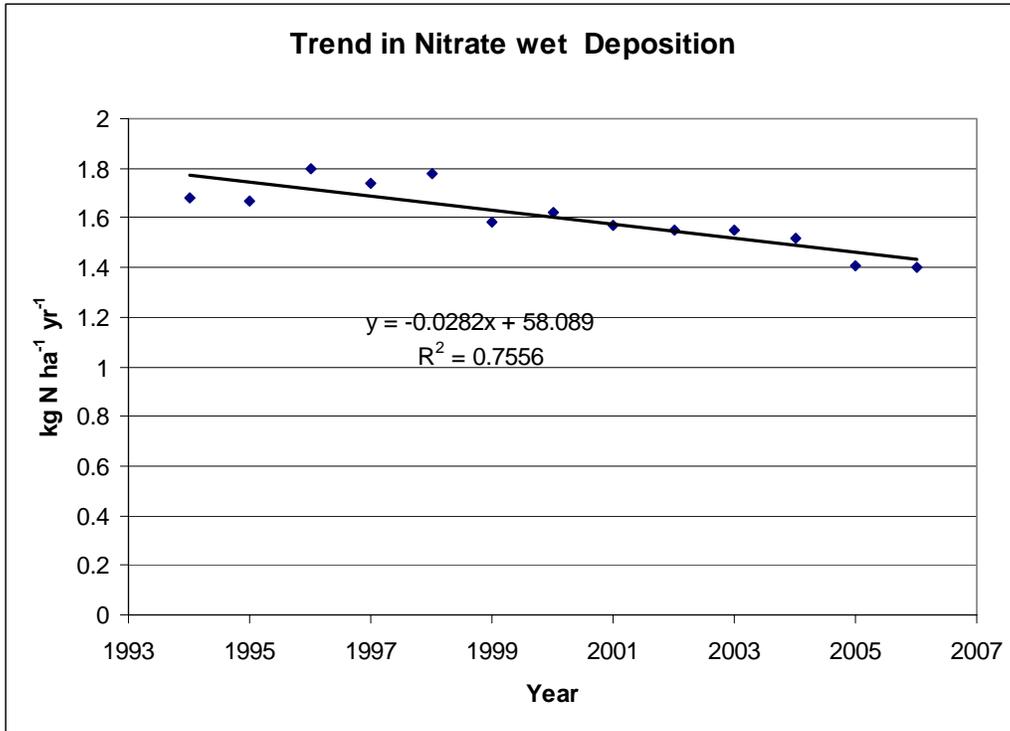
	NADP/NTN deposition estimates			
	reduced N in precipitation, kg/ha/yr	oxidized N in precipitation, kg/ha/yr	total wet N deposition, kg/ha/yr	
34				
35				
36				
37				
38	1994	1.49	1.68	3.17
39	1995	1.63	1.67	3.30
40	1996	1.66	1.80	3.45
41	1997	1.49	1.74	3.24
42	1998	1.72	1.78	3.49
43	1999	1.46	1.58	3.04
44	2000	1.48	1.62	3.10
45	2001	1.50	1.57	3.07
46	2002	1.59	1.55	3.14
	2003	1.72	1.55	3.27
	2004	1.70	1.32	3.22
	2005	1.65	1.41	3.06
	2006	1.65	1.40	3.05



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Figure 3-9. Percent change in relative contribution of oxidized and reduced 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 nitrate wet deposition, but not in ammonium deposition (Figure 3-9 and Table 3-10). 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 3-10 plotted in Figure 3-10) 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.



1
2 **Figure 3-10.** Trend in reported wet deposition of nitrate for the 48 contiguous states; data were
3 taken from Table 3-10. Note the sampling methods and locations have not been tested for
4 temporal or spatial bias.

5
6 *How is Nr Deposition related to emissions?* The relationship between emissions of Nr and
7 observed deposition is critical for understanding the efficacy of abatement strategies as well as
8 for partitioning local and large-scale effects of emissions. Only a few studies covering several
9 individual sites have sufficient monitoring consistency and duration to determine rigorously
10 long-term trends in nitrate and ammonium and their relationship to emissions, and here we
11 consider several examples [Butler et al., 2005; Kelly et al., 2002; Likens et al., 2005]. These
12 sites tend to be in the eastern US where monitoring is more concentrated and has a longer history
13 and where upwind sources and downwind receptors are relatively well known. Examination of
14 these studies reveals that concentrations of gaseous and particulate N species in the atmosphere,
15 as well as the nitrogen content of precipitation over the eastern US shows significant decreases.
16 Correlation with regional emissions is stronger than with local emissions, in keeping with the
17 secondary nature of the major compounds – nitrates and ammonium. Decreases in ammonium
18 concentration and wet deposition are attributed to decreases in sulfate concentrations meaning
19 that more of the reduced N remains in the gas phase. For the period 1965 to 2000, nitrate levels
20 in bulk deposition correlate well with reported NO_x emissions. For shorter and earlier time
21 periods the correlation is weaker, and the authors attribute this to changes in the EPA's methods
22 of measuring and reporting emissions; they find evidence of continued errors in emissions from
23 vehicles. Comparing emissions to measured ambient concentrations or flux indicates that
24 efficiency is less than quantitative, for example a 50% reduction in NO_x emissions is likely to
25 produce a reduction of about 35% in concentration and deposition of nitrate.
26

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

8
 9 A recent review [*Sickles and Shadwick, 2007a; Sickles and Shadwick, 2007b*] analyzes the
 10 seasonal and regional behavior of concentration and deposition of a variety of primary and
 11 secondary pollutants including reactive N and investigated trends from 1990 to 2004 for the US
 12 east of the Mississippi River. The investigators evaluated observations from more than 50 sites
 13 in the eastern States and concluded that for 2000-2004, the mean annual total measured N
 14 deposition for this area was 7.75 kg N ha⁻¹ yr⁻¹; see Table 3-11. This value includes vapor phase
 15 HNO₃, particulate NO₃⁻, and NH₄⁺; it does not include deposition of other oxidized species such
 16 as NO_x and PAN, nor gas-phase reduced N species most notably NH₃. The measured deposition
 17 rates peak in spring and summer, but unaccounted for ammonia deposition is probably a
 18 substantial fraction of the total, and the true annual cycle remains uncertain.

19
 20 **Table 3-11.** *Deposition of nitrogen to the eastern US in units of kg N ha⁻¹ yr⁻¹. Data are from the*
 21 *US CASTNET program for the period of 2000-2004. Monitored species for 34 sites east of the*
 22 *Mississippi include vapor-phase HNO₃, particulate NO₃⁻, and NH₄⁺; unmonitored are other*
 23 *oxidized species such as NO_x and PAN and gas-phase reduced N species most notably NH₃*
 24 *[Sickles and Shadwick, 2007a]. For an explanation of how deposition of unmeasured species*
 25 *was estimated see text.*

	Annual deposition kg N ha ⁻¹ yr ⁻¹
Dry NH ₄ ⁺	0.41
Wet NH ₄ ⁺	2.54
Dry HNO ₃ + NO ₃ ⁻	1.88
Wet HNO ₃ + 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

27
 28
 29 *Estimated Total N Deposition to the Eastern US.* CASTNET monitors nitric acid and nitrate, but
 30 not other members of the NO_y family – notably NO_x. Dennis ([USEPA, 2007](#)) estimated that the
 31 unmeasured NO_y species account for about 50% of the dry deposition of nitrates. Half of 1.88
 32 (see Table 3-11) is 0.94 kg N ha⁻¹yr⁻¹. Ammonia is also unmeasured by CASTNET, and model
 33 estimates [*Mathur and Dennis, 2003*] of NH₃ indicate that dry deposition should account for

1 75% of wet NH_4^+ deposition; 75% of 2.54 is 1.9 $\text{kg N ha}^{-1}\text{yr}^{-1}$. Adding these two values to the
2 total from Table 3-11 yields a reasonable estimate, within about $\pm 50\%$ absolute accuracy, of total
3 deposition of about 10.6 $\text{kg N ha}^{-1}\text{yr}^{-1}$ for the eastern US.

4
5 *Characteristics of N Deposition to the Eastern US.* Highest N deposition occurs in the spring
6 and summer, when chemical thermodynamics and photochemistry are conducive to removal
7 from the atmosphere. As temperatures warm, HNO_3 formation accelerates and fertilizer
8 application to agricultural fields steps up. Dry deposition for gases is faster than for particles; for
9 example the mean CASTNET reported HNO_3 deposition velocity is 1.24 cm s^{-1} while that for
10 particulate NO_3^- is 0.10 cm s^{-1} . Conversion of condensed ammonium nitrate to gaseous
11 ammonia and nitric acid is favored at high temperatures. Oxidation of NO_x to HNO_3 is faster in
12 the spring and summer due to greater ozone and OH concentrations than in the winter. Warm
13 temperatures favor release of NH_3 from soils, and summer months are the season of fastest
14 conversion of SO_2 into sulfate – ammonia combines rapidly with sulfate that is then washed out
15 of the atmosphere.

16
17 Wet deposition of ammonium and nitrate dominates deposition, averaging for the sum of
18 ammonium and nitrate $5.46 \text{ kg N ha}^{-1}\text{yr}^{-1}$, or 70 % of the total, but dry deposition cannot be
19 neglected; it averaged $2.29 \text{ kg N ha}^{-1}\text{yr}^{-1}$ or 30 % of the measured total. Because foliar resistance
20 to nitric acid is weak, dry deposition of nitrate accounts for 39 % of the total nitrate deposition.
21 When we add estimated NO_x and NH_3 dry deposition (Table 3-11), the sum of 0.41, 1.88, 0.94,
22 and 1.90 is $5.13 \text{ kg N ha}^{-1}\text{yr}^{-1}$ and rivals that delivered in precipitation.

23
24 The regional gradient is relatively modest, with the least annual average N deposition occurring
25 in the Southeast ($6.77 \text{ kg N ha}^{-1}\text{yr}^{-1}$) and the greatest in the Midwest ($8.74 \text{ kg N ha}^{-1}\text{yr}^{-1}$). These
26 gradients are driven primarily by differences in abundance – the annual mean concentration of
27 total measured atmospheric N was $1.68 \mu\text{g m}^{-3}$ in the Southeast and $2.40 \mu\text{g m}^{-3}$ in the Midwest.
28 Because only the secondary products of primary pollutants were measured, such relatively
29 uniform concentrations are typical. Also contributing to the relative spatial uniformity of
30 deposition is the greater rate of precipitation in the Southeast. The policy-relevant implication of
31 the large spatial scale nature of N deposition is that large-scale control measures are called for.

32
33 Trends in measured and inferred deposition over the 15 yr monitoring period (Figure 3-10)
34 reflect trends in emissions. In 2003 and 2004 substantial reductions in emissions from electric
35 generating units (power plants) were implemented under the NO_x State Implementation Plan
36 (SIP) call. Many of these power plants are located along the Ohio River generally upwind of the
37 measurement area. The observed trend between 1990 and 1999 was weak, but significant
38 reductions ($p = 0.05$) were found between the 1990-1994 and 2000-2004 periods [Sickles and
39 Shadwick, 2007a]. The concentration of nitric acid fell from 1.99 to $1.74 \mu\text{g N m}^{-3}$ or by 13%,
40 and total nitrate deposition fell by $0.56 \text{ kg N ha}^{-1}\text{yr}^{-1}$ or 11%. NO_x emissions controls are
41 implemented primarily in the ozone season (May to September) and greatest reductions in N
42 deposition were observed in the summer. For ammonium, the average concentration fell from
43 1.83 to $1.61 \mu\text{g N m}^{-3}$ probably as a result of lower sulfur emissions. No change was observed in
44 wet ammonium deposition.

45

1 Sickles and Shadwick [2007b] attributed the reduction in nitrate deposition to reductions in NOx
 2 emissions. They also reported that the relationship between emissions and deposition was less
 3 than 1:1. In other words, emissions were reduced by about 22%, but deposition fell by only about
 4 11%. This nonlinearity may be a function of the time intervals chosen. The second five-year
 5 period averages from 2000-2004, but reductions went into effect over the 24-month period 2003-
 6 2004. Deposition depends on both chemistry and climate, and weather shows substantial
 7 interannual variability.

8
 9 *Uncertainty in Measured Deposition.* Analysis of uncertainties in the deposition of Nr is
 10 challenging. The coefficient of variation for total, regional N deposition for 2000-2004 is 23%,
 11 representing a minimal value of uncertainty. Concentrations of some of the NOy species are
 12 monitored, as is the wet deposition of major oxidized and reduced N species, but concentrations
 13 of ammonia and other Nr species are not monitored. The network for monitoring dry deposition
 14 is spatially sparse. The monitors are located in flat areas with uniform surfaces – advective
 15 deposition into for example the edges of forests are estimated to contribute substantially to the
 16 uncertainty [Hicks, 2006]. Other sources of error include the model used to convert weekly
 17 average concentrations and micrometeorological measurements into depositions. Precision can
 18 be determined from collocated sites and is estimated at 5% for nitrate and 15% for ammonium in
 19 precipitation [Nilles *et al.*, 1994]. The uncertainty in estimated dry deposition arises primarily
 20 from uncertainty in deposition velocities [Brook *et al.*, 1997; Hicks *et al.*, 1991] and can be as
 21 high as 40% for HNO₃. Total uncertainty for deposition of Nr based on measurements is at least
 22 25% and may be as high as 50%.

23
 24 *Deposition Estimates from Numerical Models.* The EPA Community Multiscale Air Quality
 25 model (CMAQ) was run for the North America at 36 km resolution (R. Dennis *et al.*, personal
 26 communication January 2008). Calculated nitrogen deposition for the 48 contiguous states
 27 (Table 3-12) was broadly consistent with direct measurements (compare to Table 3.3.2). **This run**
 28 **of CMAQ did not account for NOx emissions from marine vessels, and these accounted for**
 29 **about 4% of the total NOx emissions in 2000.** CMAQ NOx emissions were 5.84 Tg N for the
 30 year 2002; of that 2.74 Tg N were deposited. This suggests that ~50% was exported – a number
 31 somewhat higher than has been reported in the literature; this discrepancy is discussed below.

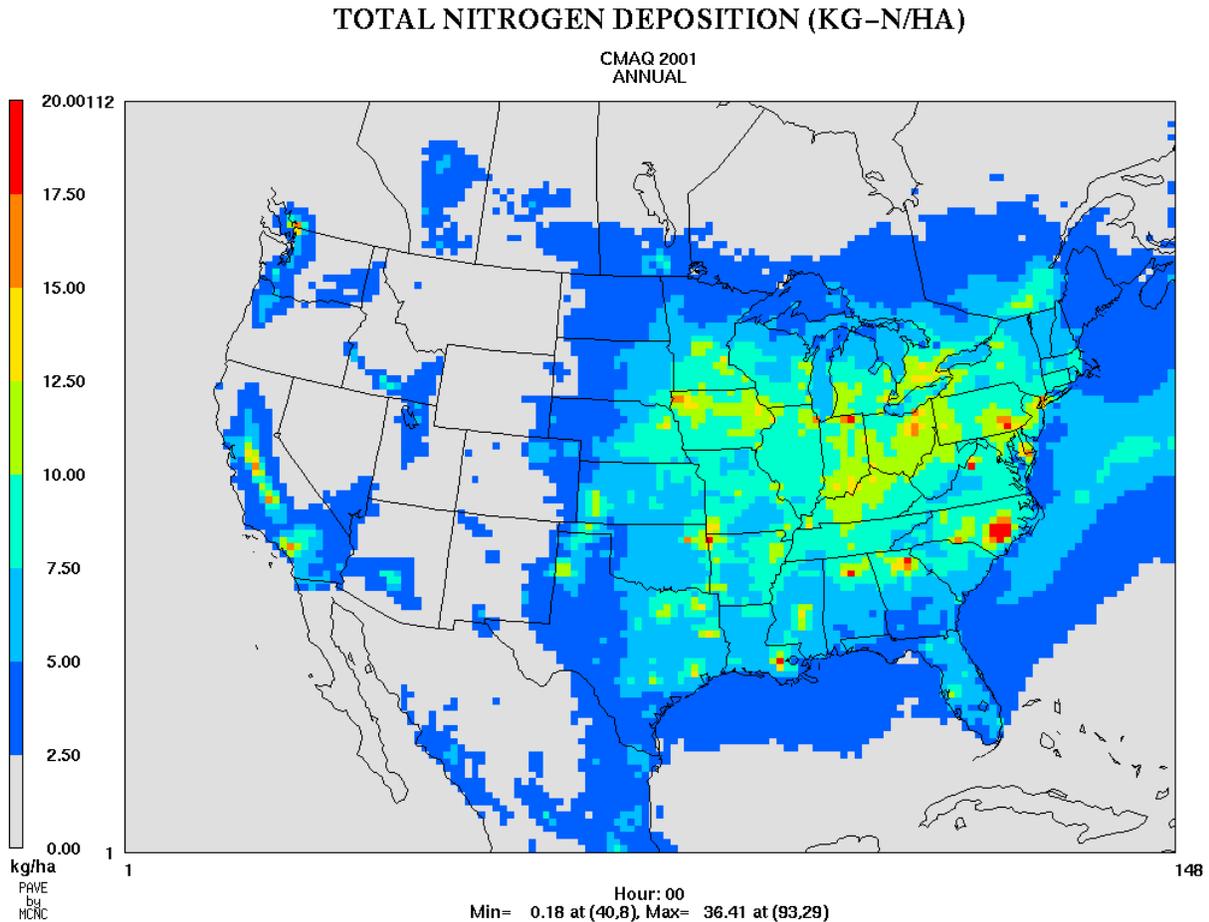
32
 33 **Table 3-12** Results from CMAQ for total deposition in 2002 to the 48 contiguous states of
 34 oxidized and reduced N.

	kg N ha ⁻¹ yr ⁻¹	Tg N yr ⁻¹
Oxidized N	3.51	2.74
Reduced N	2.66	2.07
Total N Depos.	6.17	4.81

36
 37
 38 Ammonia emissions and ambient concentrations can be measured, but are not routinely
 39 monitored. For reduced nitrogen, the CMAQ numerical simulation employed inverse modeling
 40 techniques – that is ammonia emissions were derived from observed ammonium wet deposition
 41 [Gilliland *et al.*, 2006; Gilliland *et al.*, 2003; Mathur and Dennis, 2003]. Model determinations
 42 therefore do not provide an independent source of information on ammonium deposition.

1 The three-year CMAQ run gives an indication of the spatial pattern of deposition and the relative
2 importance of wet and dry deposition (Figures 3-11 and 3-12). For NH_x , wet and dry are
3 equally important, but for NO_y , dry deposition is greater than wet. While this is not true for the
4 eastern US it is true for the US as a whole; in arid southern California, for example, dry
5 deposition of Nr dominates. Based on CMAQ, total NO_y deposition is 2.79 times the wet
6 deposition and total NH_x deposition is 1.98 times the wet deposition. Using the data from Table
7 3-12 for the average wet deposition for the period 2000- 2004, total deposition of oxidized N is
8 $4.36 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ ($2.79 * 1.56 = 4.36$). The total deposition for reduced N is $3.17 \text{ kg N ha}^{-1} \text{ yr}^{-1}$
9 $(1.98 * 1.60)$. The grand total (wet and dry oxidized and reduced) is then about 7.5 kg N ha^{-1}
10 yr^{-1} .

11
12

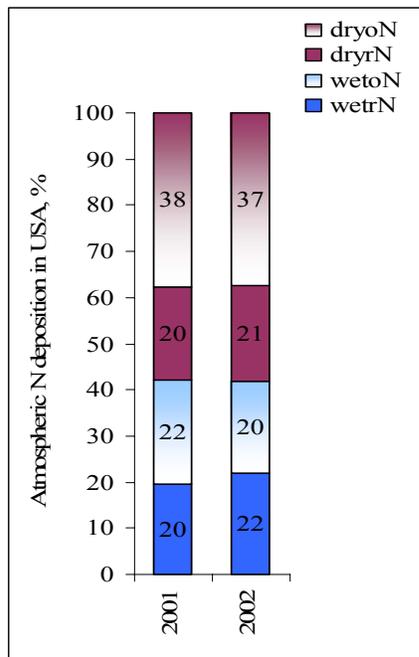


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Figure 3-11. CMAQ annual average (wet plus dry and oxidized plus reduced) nitrogen deposition (in $\text{kg-N ha}^{-1} \text{ yr}^{-1}$) across the U.S. based on 3 years of differing meteorology - one dry, one wet, and one average precipitation year - across the Eastern U.S. (Source: US EPA, 2007). 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 NO_x emissions from point sources over the eastern US.

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Figure 3-12. *Relative contributions of wet and deposition for reduced and oxidized N. Results CMAQ runs for the 48 contiguous States.*



*dry
from*

For comparison purposes, a collection of Chemical Transport Models (CTM's) [Dentener *et al.*, 2006] yielded total (wet plus dry) deposition to the whole US of about 3.9 Tg N yr⁻¹ oxidized reactive N and 3.0 Tg N yr⁻¹ ammoniacal N for current emissions. The fate of NO_x is assumed to be primarily nitric acid or aerosol nitrate – 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⁻¹. The variance among models was about 30% (one σ) 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 ammonia, 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 atmospheric chemically reactive N (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⁻¹ 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

1 winter; this represents about 6.5% of the total N emissions at the time. More recent estimates,
2 again using data from lower to mid tropospheric altitudes over the eastern US [Li et al., 2004;
3 Parrish et al., 2004b], estimated that 10 - 15% of the emitted NO_x was exported in the spring
4 and fall. A summer season determination [Hudman et al., 2007] indicated about 15% NO_x
5 export in the 2.5–6.5 km altitude range.

6
7 None of these studies, based on observations or combinations of observations and models,
8 evaluated N flux resulting from deep convection, which can account for substantial transport of
9 boundary layer (BL) air in the summer [Chatfield and Crutzen, 1984; Luke et al., 1992; Ryan et
10 al., 1992]. Uncertainty in the convective mass flux and in NO produced by lightning make direct
11 determination of NO_y vented from the BL difficult. The convective mass flux is at present a
12 poorly constrained quantity, uncertain to about a factor of two [Doherty et al., 2005; Lawrence et
13 al., 2003].

14
15 In an early, model-based mass balance study [Kasibhatla et al., 1993], wet and dry deposition in
16 source regions were estimated to account for 30% and 40-45% of the emissions, respectively.
17 The authors reported that the remainder (25-30%) was exported off the continent, and more
18 recent modeling studies tend to agree with a determination of 65-75% deposition [Doney et al.,
19 2007; Galloway et al., 2004; Holland et al., 1997; Horowitz et al., 1998; Liang et al., 1998]. In
20 general, these CTM's derived small export values – on the order of 30% of the total NO_x emitted
21 into the lower atmosphere. For example, Park et al. (2004) used a stretched-grid global model
22 with highest resolution over the US to estimate NO_x and NO_y export for June 1985. They
23 reported boundary layer NO_y advection of 0.56 Tg N yr⁻¹ and total exports of 1.94 Tg N yr⁻¹;
24 deposition accounted for ~76% of the emitted NO_x. There is substantial model-to model
25 variability within one model [Penner et al., 1991] putting more nitrate deposition into the Gulf of
26 Mexico. The models appear to match well the measured boundary layer export and the ratio of
27 NO_x/NO_y, e.g., [Luke et al., 1992; Parrish et al., 2004a] and generally agree with direct
28 measurements. In summary, reviewed publications using the mass balance approach have
29 substantial uncertainty but indicate with some consistency that 25-35% of the NO_y emitted over
30 the US is exported.

31
32 *Comparison of models and measurements of oxidized N deposition.* Both ambient measurements
33 and numerical models of NO_y have reached a level development to allow reasonable estimates of
34 deposition. For reduced nitrogen, neither ambient concentrations nor emissions are known well
35 enough to constrain models. Here we will review published research on NO_y export and
36 deposition. Recent model estimates of the US N budget are reasonably uniform in finding that
37 about 25-35% of total NO_x emissions are exported. From those studies we can estimate the
38 vertical flux into the surface of the 48 contiguous states. For the 2000-2002 period, total NO_x
39 emissions were about 4.5 Tg N yr⁻¹. The upper limit to deposition, if all of this is deposited onto
40 the continent, would have been 5.7 kg N ha⁻¹yr⁻¹ for the 7.8 x 10⁶ km² (7.8 x 10⁸ ha) surface area
41 of the 48 contiguous States. The studies reviewed above suggest that 70% of the N released is
42 deposited, and this works out to ~4.0 kg N ha⁻¹yr⁻¹. This is comparable to the oxidized N
43 deposition of 5.7 kg N ha⁻¹yr⁻¹ estimated from CASTNET observations for the eastern States
44 (Table 3-10).

45

SAB Draft Report to Assist Meeting Deliberations -- Do not Cite or Quote

This draft is a work in progress, does not reflect consensus advice or recommendations, has not been reviewed or approved by the chartered SAB, and does not represent EPA policy

1 Results from CMAQ runs, described above, indicate that of the NO_x emitted over the continental
2 US 50% is deposited and 50% is exported. This is within the combined error bars of other
3 studies, but well under the best estimate of 70% deposition. One possible source of this
4 discrepancy is organo-nitrogen compounds. The mechanism for formation and deposition of
5 organic nitrates is uncertain, and the chemical mechanism used in CMAQ was highly simplified
6 – only about 2-3% of the total Nr deposition can be attributed to organo-nitrogen compounds (R.
7 Dennis personal communication, 2008). Duce et al (2008) suggest that organic Nr constitutes a
8 fair fraction of the Nr load. Many of these compounds (such as peroxy-methacrylic nitric
9 anhydride, CH₂C(CH₃)C(O)OONO₂) are formed by reactions between VOC's and NO_x. Such
10 compounds are detected as NO_y and are thus included in measurements of Nr export. Arbitrarily
11 up-scaling of CMAQ deposition would then violate mass balance. EPA should investigate the
12 source of this discrepancy and support research to reduce the uncertainty in Nr deposition and
13 export.

14

15 The total wet deposition of nitrate to the 48 contiguous states averaged 1.6 kg N ha⁻¹yr⁻¹ for the
16 period 2000-2002 (Table 3-11). If we assume an equal amount is lost from the atmosphere
17 through dry deposition, then the total deposition of oxidized N to the surface is 3.2 kg N ha⁻¹yr⁻¹,
18 close to the implied model results of ~4.0 kg N ha⁻¹yr⁻¹. The estimate of equal fractions wet and
19 dry deposition carries substantial uncertainty – NADP maps show, for example, little wet
20 deposition of nitrate in southern California, but this region is known to experience high
21 concentrations of NO_y. Neither approach to determining deposition is certain to better than
22 about ± 50%, so additional work is called for.

23

24 Major sources of uncertainty in modeled and observed values include missing deposition terms
25 and poorly constrained convective mass flux. As indicated above, convective mass flux (rapid
26 vertical transport) is uncertain because most convective clouds are smaller than a grid box in a
27 global model. There is evidence for nonlinearities in NO₂ deposition velocities with greater
28 transfer from the atmosphere to the surface at higher concentrations [*Horii et al.*, 2004; , 2006].

29

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

36

37 1. Measurements from the National Atmospheric Deposition Program (NADP) indicate that
38 wet deposition of ammonium plus nitrate for the period 2000 – 2006 averaged 3.1 kg N ha⁻¹
39 yr⁻¹ over the 48 contiguous States.

40

41 2. The reduced and oxidized forms of reactive N contributed about equally to the flux, but
42 input to the eastern US was greater (and less uncertain) than to the western US.

43

44 3. For the US east of the Mississippi, dry deposition data have also been analyzed – the
45 Clean Air Standards and Trends Network (CASTNET) monitors vapor phase HNO₃, as well
46 as particulate NO₃⁻ and NH₄⁺. These measurements indicate 7.75 kg N ha⁻¹yr⁻¹ total

1 deposition (5.46 wet 2.29 dry) over the East. Conspicuous by its absence from this number
2 is dry deposition of ammonia.

3
4 4. Reductions in NO_x emissions appear to lead to reductions in deposition. NADP data
5 show a national decreasing trend in the wet nitrate deposition and some individual sites
6 show statistically significant reductions in deposition and correlations with emissions.

7
8 5. A thorough review of all published studies of the US NO_y budget indicates that about 70
9 % of the NO_x emitted by the US is deposited onto the continent with the remainder
10 exported, although substantial uncertainty remains. Major sources of error include dry
11 deposition of unmonitored members of the NO_y family, uncertainties in the chemistry of
12 organic nitrogen, and poorly constrained estimates of convective venting of the planetary
13 boundary layer.

14
15 6. Based on observations and model estimates of the relative deposition of unmeasured
16 quantities, total estimated deposition of all forms of reactive N for the period 2000-2004 is
17 ~11 kg N ha⁻¹yr⁻¹ for the eastern US, and for the 48 States ~7.5 kg N ha⁻¹yr⁻¹ with a range of
18 5.5 to 9.5 kg N ha⁻¹yr⁻¹.

19
20 **Finding F3-9.** Scientific uncertainty in origins, transport, chemistry, sinks, and export of reactive
21 nitrogen remains high, but evidence is strong that atmospheric deposition of Nr to the Earth's
22 surface as well as emissions from the surface to the atmosphere contribute substantially to
23 environmental and health problems.

24
25 **Finding F3-10.** Emissions of reactive nitrogen from agricultural practices (primarily in the form
26 of NH₃) have not been well monitored, but appear to be increasing. Both wet and dry deposition
27 contribute substantially to NH_x removal, but only wet deposition is known with much scientific
28 certainty. Ammonia, ammonium, and possibly organic nitrogen levels in the atmosphere are too
29 high to protect public health and welfare, and reductions of NH_x emissions are necessary.

30
31 **Scientific Research Recommendations:**

32 **Recommendation R3-9.** Increase the scope and spatial coverage of the Nr concentration and
33 flux monitoring network (such as CASTNET) for the US and appoint an oversight panel.

34
35 **Recommendation R3-10.** Monitor NH₃, NH_x, NO_y, NO₂, NO, and PAN concentrations,
36 measure or infer deposition, and support the development of new measurement and monitoring
37 techniques.

38
39 **Recommendation R3-11.** The current NO₂ standard is inadequate to protect health and welfare,
40 and compliance monitoring for NO₂ is inadequate for scientific understanding.

41
42 **Recommendation R3-12.** Measure deposition directly both at the CASTNET sites and nearby
43 locations with non-uniform surfaces such as forest edges.

44
45 **Recommendation R3-13.** Continued and support research into convective venting of the PBL
46 and long range transport.

1
2 **Recommendation R3-14.** Develop techniques and support observations of atmospheric organic
3 N compounds in vapor, particulate, and aqueous phases.

4
5 **Recommendation R3-15.** Improve quality and quantity of measurements of the NH₃ flux to the
6 atmosphere from major sources especially agricultural practices.

7
8 **Recommendation R3-16.** Improve numerical models of NO_y and NH_x especially their
9 chemistry, surface deposition, and export. Develop linked ocean-land-atmosphere models of Nr.

10
11 *3.3.1.2 Input and recycling of Nr within terrestrial systems in the USA*

12
13 This section builds upon Section 3.2 by integrating the information in that section on Nr
14 introduction into the US and its loss to environmental systems by energy and food production
15 into the overall picture of Nr cycling within terrestrial systems.

16
17 Annual input of newly created Nr onto terrestrial ecosystems comes from atmospheric
18 deposition, synthetic fertilizer and BNF in managed and unmanaged ecosystems (Table 3.1.).
19 Although Nr from atmospheric deposition is formed inadvertently during fossil fuel combustion
20 and from volatilization of ammonia from agricultural activities it serves to provide fertilizer,
21 along with biological N fixation and synthetic fertilizer, for food, feed and fiber production in the
22 agricultural sector. Forests and grasslands use Nr for growth and home gardens, parks and
23 recreational areas utilize Nr within the urban landscape. Approximately 32 Tg of new Nr
24 reached the land of the 48 contiguous states in 2002 (Table 3.1). An additional ~0.2 Tg of N was
25 imported mainly as food and drink products (FAO, 2008). An additional ~12 Tg of Nr was
26 recycled back to terrestrial systems in livestock (~6 Tg N) excreta, human (~2 Tg N) excreta, and
27 crop residue from the previous year's production (~4 Tg N; USEPA, 2007). Of this N ~ 1.3 Tg
28 (~1.2 from livestock manure and <0.1 from sewage sludge) was used as fertilizer for crop
29 production (USEPA, 2007).

30
31 Most of the new Nr (~17 Tg total with 9 from synthetic fertilizer and ~8 from biological N
32 fixation; Table 3-13) was used to produce food for human consumption and forage and feed for
33 livestock and poultry. In addition to new Nr and Nr that was recycled from livestock and human
34 excreta, crop production releases Nr that is stored in soil organic matter (see section 3.3.2). The
35 N in cereal crops is typically derived from added fertilizer (synthetic or manures) and from
36 mineralization of soil organic matter (conversion of complex organic molecules to ammonium)
37 in about equal amounts. As discussed in Sections 3.2 and 3.3.5.1, crop production is not efficient
38 in using Nr so only 30-70% (a global average of 40%) of all of the N mobilized for crop
39 production is harvested in the crop. The remainder is stored in the soil, leached to aquatic
40 systems as nitrate, volatilized to the atmosphere as NH₃ or NO_x or denitrified (see Section. 3.4)
41 to produce NO_x, N₂O and N₂. An additional ~1.1 Tg of synthetic fertilizer N is used to maintain
42 turfgrass in the urban environment (see Section 3.2.4) and another 0.1-0.2 Tg N is used to
43 enhance forest production.

Table 3-13 Sources of reactive N input into terrestrial systems in the USA in 2002 (from Table 3.1).

Source	-----Environmental System-----				Total
	Agricultural	Vegetated		Populated	
		Forest	Grassland		
		-----Tg N-----			
Atmospheric N Fixation	1.3	1.4	1.9	0.4	6.9*
Synthetic N	7.7	--	6.4	--	13.7
Animal Manure	9.7	0.1	**	1.1	10.9
Human Sewage	1.2	--	3.8#	--	6.0#
	0.1	--	--	1.2	1.3
Total Nr Input	20.0	1.5	5.7	3.4	38.8

**Total area does not sum to 100% because non arable lands are not included*
***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 (USEPA, 2007).

Within the N Cascade (Fig. 3-1) 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. Nr input into these systems is from atmospheric deposition and Nr from livestock manure that is deposited while the livestock is grazing within grasslands (Fig. 3-13) may lead to the N saturation of unmanaged forest and grassland ecosystems (Galloway et al. 2004).

We will now use the Nr input numbers from Table 3-13, and food production numbers to estimate the flow of Nr through agricultural and populated parts of the terrestrial system (Figure 3-13). The FAO (2008; www.fao.org/statistics/toptrade/trade.asp) lists the 20 largest agricultural commodities produced, imported and exported in the USA 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 USEPA (2007) 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), 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 nitrogen input use efficiency is 40% then ~23 Tg of N from all sources is required to produce 9.3 Tg of vegetative commodities. Table 3-14 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

1 then 24.3 Tg of N was required to produce the 9.3 Tg of crop commodity N. These estimates
 2 indicate that ~38% of the total annual input of N that went into the agricultural crop production
 3 system was contained in the main crop commodities produced in the USA in 2002.

4
 5 Of this 24.3 Tg N approximately 2.5 Tg was used to grow feed used for milk, egg and meat
 6 production. This estimate is made assuming that 4 units of N are required to produce a unit of
 7 milk, eggs or meat (see section 3.3.5.1.). This estimate also assumes that 1/3 of N required for
 8 livestock production comes from commodities in the FAO top 20 list and the remaining 2/3
 9 comes from alfalfa, silage and grass over the course of a year (Oitjen and Beckett, 1996)
 10 Approximately 4.3 Tg of N in agricultural commodities (2.8 Tg in soybeans, corn and wheat)
 11 were exported while ~0.15 Tg N was imported in various food and drink commodities. The
 12 USA human populace consumed ~1.96 Tg of N in 2002 (292 million people, consume 114.7 g
 13 protein person⁻¹ day⁻¹, 0.16 g N g⁻¹ protein⁻¹, 365 days) (approximately 1.2 Tg from animal
 14 protein-N and 0.7 from vegetative protein).
 15

16 **Table 3-14. Total N sources for crop production in 2002. The major USA crops contained**
 17 **9.3 Tg N (FAO, 2008).**

18 N Source	Tg N
19 Atmospheric	1.3
20 N Fixation	7.7
21 Synthetic N	9.7
22 Animal Manure	1.2
23 Sewage Sludge	0.1
24 Soil N	4.7
25 Crop Residue*	4.4
26	
27	
28 Total	29.1
29	

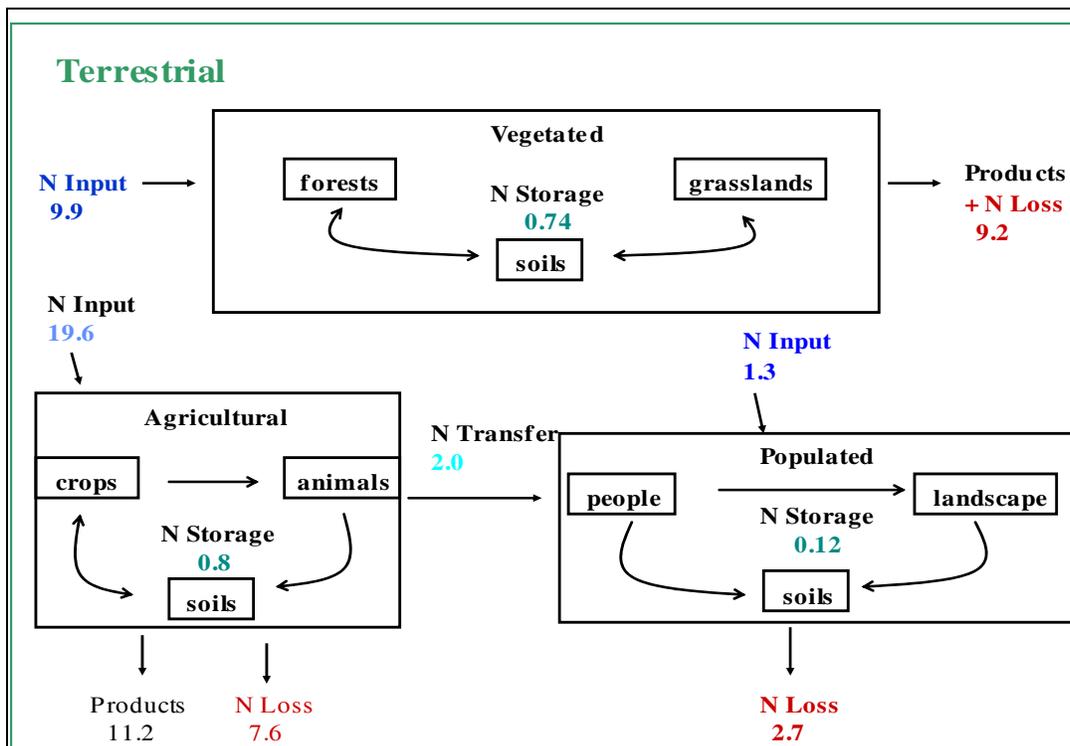
30 *Using EPA 2007 estimate

31
 32 These three consumption areas, internal consumption of vegetable N for livestock production,
 33 human consumption and export account for 77% of the commodities produced. The
 34 unaccounted for commodity N is likely partly in annual storage. Some smaller fraction of annual
 35 production is used for pet food and a small fraction is returned to the terrestrial environment
 36 because of spoilage and handling losses.
 37

38 In forests and grasslands (vegetated system) N input in 2002 was ~3.5 Tg of anthropogen ically
 39 introduced N. Of this N ~21% was retained in soil and tree biomass while the remainder was
 40 removed in tree harvest or lost to other parts of the environment through ammonia volatilization
 41 and nitrate leaching and runoff (Fig. 3-13). Total N input into agricultural systems was ~19.6 Tg
 42 with ~ 11 Tg being removed as products which includes the transfer of ~2 Tg N as food to the
 43 human population. Almost 40% of the N input into agricultural systems is lost through ammonia
 44 volatilization, nitrification/denitrification and nitrate leaching and runoff. The 4.2 Tg of Nr of
 45 Haber-Bosch N that is used for industrial feedstock is not included in this assessment. Of the

1 input of ~3.3 Tg of N into the populated system ~80% is lost through human excreta processed
 2 in sewage treatment plants, denitrification in soils and leaching and runoff of nitrate (Fig. 3-13).
 3

4 Figure 3-13 summarizes the input and flow of Nr in the main terrestrial systems within the
 5 continental U.S. Anthropogenic input of Nr into forests and grasslands totaled ~3.5 Tg in 2002
 6 with an estimated 6.4 Tg of Nr being introduced through natural biological N fixation. Of this
 7 Nr ~ 0.7 Tg was stored in vegetation and soils (see section 3.3.2) while the remainder lost to the
 8 atmosphere and aquatic systems, or removed as forest products and livestock forage. The largest
 9 anthropogenic Nr input (~19.6 Tg) was into agricultural production where ~11.2 Tg was
 10 removed as agricultural product, ~ 2 Tg transferred as edible product to the “populated” portion
 11 of the terrestrial system, ~0.8 Tg was stored in Agricultural lands, and ~7.6 Tg N was lost to the
 12 atmosphere and aquatic systems. New N input into the “Populated” portion totaled ~3.3 Tg,
 13 which came from N transfer in food and use of fertilizer N in lawns, gardens and recreational
 14 areas. Within these areas an estimated 0.12 Tg was stored in urban forests.
 15
 16



17
 18
 19
 20 **Figure 3-13. The terrestrial portion of the Nitrogen Cascade for the continental USA in**
 21 **2002. Numbers are Tg Nr/yr.**
 22

23
 24 *3.3.1.3 Transfer of Nr to Aquatic Systems*
 25

26 Within the Nitrogen Cascade, Nr flows from the atmosphere and terrestrial systems into aquatic
 27 systems. Aquatic systems include groundwater, wetlands, streams and rivers, lakes and the

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1 coastal marine environment. Nr is deposited directly into surface aquatic systems from the
2 atmosphere (direct deposition) and Nr that is not either stored or removed as products on
3 terrestrial systems eventually moves into aquatic systems (indirect deposition). What is the
4 concern about too much Nr in aquatic systems? The USEPA Office of Water notes the following
5 reasons for implementing numerical water quality standards:

6
7 *Excessive nutrients (nitrogen and phosphorus) can cause negative ecological impacts to water bodies on
8 a national scale by stimulating harmful algal blooms.*

9 *Algal blooms block sunlight and result in the destruction of submerged aquatic vegetation which
10 serves as critically important habitat and food for many organisms.*

11
12 *Algal blooms eventually die off and consume dissolved oxygen from the water
13 column which can lead to die off of aquatic organisms.*

14
15 *One result of algal blooms is decreased biological diversity and populations, including
16 smaller populations of game and commercial fish.*

17
18 *Some blooms, considered “harmful algal blooms” or “HABs”, have a toxic effect on living
19 organisms and are disruptive of ecosystem structure and transfer of energy to higher trophic
20 levels.*

21
22 *Excessive nutrients also pose public health risks.*

23 *Algal blooms can cause taste and odor problems in drinking water.*

24 *Hazardous algal blooms can cause respiratory distress and neurological problems in
25 swimmers.*

26
27 *Excessive nitrates can cause blue baby syndrome.*

28
29 *Nutrient pollution is occurring at a national scale and has not been completely addressed .*

30
31 *49 states and 4 territories have 303(d) listings due to nutrients, and about 50% of the
32 states have greater than 100 water quality impairments due to nutrients .*

33
34 *Over 10,000 impairments are a result of nutrient pollution .*

35
36 Mitsch et al. (2001) suggest that streams and rivers themselves are not always much affected by
37 nutrient loading as are lakes, wetlands, coastal areas and other lentic bodies of water. However,
38 in most cases these nutrient-enriched waterways flow to the sea, with eutrophication of coastal
39 waters the unfortunate result. This problem now occurs regularly throughout the world (WRI,
40 2007), in locations such as the Gulf of Mexico (Rabalais et al. 1996), the Baltic Sea (Larson et al.
41 1985), and the Black Sea (Tolmazin 1985).

42
43 The availability of nitrogen (N) controls or “limits” primary production in much of the world’s
44 estuarine, near-shore coastal and open ocean waters (Dugdale 1967, Ryther and Dunstan 1971,
45 Nixon 1995, Smetacek et al. 1991, Jørgensen and Richardson 1996; Duce et al., 2008). Nitrogen
46 can also play a role as either a primary or secondary limiting nutrient in freshwater
47 environments, especially large lakes (e.g. L. Tahoe, L. Superior). As such, the fertility of these
48 waters is often closely controlled by N inputs. Nitrogen inputs are provided either internally by
49 regeneration of pre-existing N and biologically-fixed atmospheric N₂, or supplied externally (i.e.

1 “new” N) as combined N sources delivered via surface runoff, sub-surface groundwater or
2 atmospheric deposition. Because marine ecosystems lose some fixed N via denitrification and
3 burial, “new” N supply is needed to compensate for these N losses. During the past century,
4 following large-scale use of synthetic N fertilizers in agriculture, rapid expansion of industrial
5 and transportation-related fossil fuel combustion and coastal urbanization, humans have
6 significantly altered the balance between “new” N inputs and N losses in the marine environment
7 (Codispoti et al. 2001, Galloway and Cowling 2002). During this time frame, terrigenous
8 discharge and atmospheric N emissions have increased by 10 fold (Howarth et al. 1996, Holland
9 et al., 1999). This number keeps growing as human development continues to expand in coastal
10 watersheds (Vitousek et al. 1997).

11
12 For at least 50 years, researchers have recognized this growing imbalance, especially in estuarine
13 and coastal waters where anthropogenically-derived N over-enrichment has fueled accelerated
14 primary production, or “cultural” eutrophication (Vollenweider et al. 1992, Nixon 1995).
15 Eutrophication is a condition where nutrient-enhanced primary production exceeds the ability of
16 higher ranked consumers and organic matter-degrading microbes to consume and process it.
17 D’Elia (1987) characterized this condition as “too much of good thing”, or over-fertilization of
18 N-limited marine ecosystems with “new” N, a bulk of it being anthropogenic (Howarth et al.
19 1986, Vitousek et al. 1997, Galloway and Cowling 2002). Symptoms of N-driven eutrophication
20 vary from subtle increases in plant production to changes in primary producer community
21 composition, to rapidly accelerating algal growth, visible discoloration or blooms, losses in water
22 clarity, increased consumption of oxygen, dissolved oxygen depletion (hypoxia), which is
23 stressful to resident fauna and flora, or in the case of total dissolved oxygen depletion (anoxia),
24 elimination of habitats (Paerl 1988, 1997, Diaz and Rosenberg 1995, Rabalais and Turner 2001).
25 Other effects include submerged aquatic vegetation (SAV) losses, possible impacts on tidal
26 wetland health, and disruption of estuarine food chain dynamics that may favor an imbalance
27 towards lower trophic levels (e.g., jellyfish) or microbial looping.

28
29 Anthropogenic or cultural eutrophication has been closely linked to population densities in
30 coastal watersheds (Peierls et al., 1991, Nixon 1995, Vitousek et al. 1997). Primary sources of N
31 enrichment include urban and agricultural land uses as well as wastewater treatment plants, most
32 of which have not been designed to remove nutrients. A significant, and in many instances
33 increasing, proportion of “new” N input can also be attributed to remote sources residing in
34 airsheds. Delivery routes can also be complex, especially when via subsurface aquifers outside
35 the immediate watershed, which can confound source definition and create long delays in
36 delivery and management response (Paerl 1997, Jaworski et al., 1997, Galloway and Cowling
37 2002, Paerl et al. 2002).

38
39 The area of an airshed generally greatly exceeds that of a watershed for a specific estuary or
40 coastal regions. For example, the airshed of the Baltic Sea includes much of Western and
41 Central Europe (Asman 1994, Hov et al., 1994), while the airsheds of the US’s two largest
42 estuarine ecosystems, the Chesapeake Bay and Albemarle-Pamlico Sound, are 15 to over 30
43 times the size of their watersheds (Dennis 1997). Thus, the airshed of one region may impact the
44 watershed and receiving waters of another, making eutrophication a regional-scale management
45 issue (Paerl et al. 2002, Galloway and Cowling 2002). Furthermore, atmospheric N inputs do
46 not stop at coastal margins. Along the North American Atlantic continental shelf, atmospheric N

1 inputs more than match riverine inputs (Jaworski et al., 1997, Paerl et al., 2002), underscoring
2 the fact that N-driven marine eutrophication may require regional or even global solutions. Even
3 in truly oceanic locations (e.g. Bermuda), North American continental atmospheric N emissions
4 (reduced and oxidized N) are commonly detected and significant (Luke and Dickerson 1987,
5 Prospero et al. 1996). Likewise, islands in the North Pacific receive N deposition originating in
6 Asia (Prospero et al., 1989).

7
8 Riverine and atmospheric “new” Nr inputs in the North Atlantic Ocean basin are at least equal
9 and may exceed “new” Nr inputs by biological N₂ fixation (Howarth et al. 1996, Paerl and
10 Whitall 1999, Paerl et al. 2002). Duce et al. (2008) estimate that up to a third of ocean’s external
11 Nr supply enters through atmospheric deposition. This deposition leads to an estimated ~ 3% of
12 new marine biological production and increased oceanic N₂O production. Therefore, our
13 understanding of marine eutrophication dynamics, and their management, needs to consider a
14 range of scales reflecting these inputs, including ecosystem, watershed, regional and global
15 levels.

16
17 *Scope of the Problem in the United States.* Over the past 25 years, there has been a growing
18 recognition of cultural eutrophication as a serious problems in coastal estuaries (NRC, 2000).
19 Globally, Selman et al. (2008) have reported, “Of the 415 areas around the world identified as
20 experiencing some form of eutrophication, 169 are hypoxic and only 13 systems are classified as
21 ‘systems in recovery’”. Comprehensive surveys of U.S. estuaries have been conducted by
22 NOAA as part of the National Estuarine Eutrophication Assessments (NEEA) in 1999 and 2004
23 (Bricker et al. 1999; 2007). The most recent report, released in 2007 (Bricker et al., 2007)
24 focused on nutrient enrichment and its manifestations in the estuarine environment and relies on
25 participation and interviews of local experts to provide data for the assessment. Among the key
26 findings for nearly 100 assessed U.S. estuaries were that eutrophication is a widespread problem,
27 with the majority of assessed estuaries showing signs of eutrophication—65% of the assessed
28 systems, representing 78% of assessed estuarine area, had moderate to high overall eutrophic
29 conditions. The most common symptoms of eutrophication were high spatial coverage and
30 frequency of elevated chlorophyll *a* (phytoplankton)—50% of the assessed estuaries,
31 representing 72% of assessed area, had a high chlorophyll *a* rating.

32
33 Further field evaluations by EPA and state and university collaborators under the National
34 Coastal Assessment (NCA) using probabilistic monitoring techniques The NCA National Coastal
35 Condition Reports (U.S. EPA, 2001, 2004 and 2006) are more closely related to nutrient
36 enrichment assessments, especially for manifestations of nutrient enrichment such as hypoxia,
37 nuisance algal blooms, and general habitat degradation. The last comprehensive national NCCR
38 was published in 2004 (U.S. EPA, 2004) with a more recent assessment focused on 28 National
39 Estuary Program estuaries published in 2007 (U.S. EPA, 2006). The 2004 NCCR included an
40 overall rating of “fair” for estuaries, including the Great Lakes, based on evaluation of over 2000
41 sites. The water quality index, which incorporates nutrient effects primarily as chlorophyll-a and
42 dissolved oxygen impacts, was also rated “fair” nationally. Forty percent of the sites were rated
43 “good” for overall water quality, while 11% were “poor” and 49% “fair”.

Hypoxia in the Gulf of Mexico Hypoxia

An example of a problem of excess Nr that moves from one part of the USA 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 USEPA, 2007a). 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 km² (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; USEPA, 2007a).

The report, Hypoxia in the Northern Gulf of Mexico. An update by the EPA Science Advisory Board. December, 2007 (USEPA, 2007a) 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 F3-11: Over the past 25 years, there has been a growing recognition of anthropogenic eutrophication as a serious problems in coastal estuaries (NRC, 2000). The last comprehensive national NCCR was published in 2004 (U.S. 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”.

Scientific Research Recommendations R3-17: Our understanding of marine eutrophication dynamics, and their management, needs to consider a range of scales reflecting ecosystem, watershed, and regional levels that include all inputs, e.g. atmospheric and riverine.

3.3.2. Storage of Nr within Terrestrial ES

According to the N 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

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1 fixation in crops (mainly soybeans), and 1.3 Tg from atmospheric deposition. Nr input into
2 vegetated systems (mostly forested, but including non-cropland grasslands and other natural
3 vegetation types as well) comes mostly from atmospheric deposition (3.2 Tg). Annual input of
4 Nr into populated systems includes synthetic fertilizer application to urban turfgrass and
5 recreational areas (~1.1 Tg), and atmospheric deposition 0.2 Tg (Table 3-13.).
6

7 Much of the annual Nr input into these terrestrial systems passes through, and is transferred
8 within, terrestrial systems or atmosphere via NH₃, NO_x or N₂O, or aquatic environmental
9 systems via nitrate and organic N leaching and runoff or NH_x and NO_y deposition.
10

11 The largest single reservoir of total N in the terrestrial environmental system is soil organic
12 matter (SOM). Approximately 52,000 Tg C and 4,300Tg N are contained in the upper 100 cm of
13 soil in the 48 contiguous states (N is estimated from assumed C/N ratio of 12) (Lal et al. 1998).
14 For comparison, the total above ground biomass of U.S. forests of these states contains ~ 15,300
15 Tg of C and ~ 59 Tg N (estimated using a C/N ratio of 261, and 15,500 Tg of SOM-C, 1290Tg
16 total N (estimated using a C/N ratio of 12) (USEPA, 2007b). Most of this SOM-N is bound
17 within complex organic molecules that remain in the soil for tens to thousands of years. A small
18 fraction of this SOM is mineralized, converted to carbon dioxide and Nr, annually. The total N
19 contained within above and below ground compartments isn't really of concern, what is of
20 interest in addressing issues of Nr, is the change in N stored within the compartments of
21 terrestrial systems. The pertinent question is, is N being retained or released from longterm
22 storage? Let us look at some estimates of annual change of N storage within important
23 components of terrestrial systems.
24

25 *3.3.2.1. Agricultural*
26

27 Croplands within the contiguous 48 states occupy ~149 million ha (19%) of the 785 million ha of
28 land area, of which 126 million ha were cultivated in 2002 (NRCS, 2007;
29 www.nrcs.usda.gov/technical/land/nrio3/national_landuse.html). Croplands are generally found
30 on well drained mineral soils (organic C content 1-6% in the top 30 cm). Small areas of drained
31 organic soils are cultivated (organic C content of 10-20%) in mainly Florida, Michigan and
32 Minnesota (USEPA, 2007a). Organic soils lost ~0.69 Tg of Nr in 2002 while mineral soils
33 accumulated ~1.5 Tg of Nr (Table 3-15). Much of the accumulation of SOC was due to the use
34 of conservation tillage and high yielding crop varieties (USEPA, 2007a). Losses of Nr from
35 organic soils are due to mineralization of SOM and release of Nr input. In cultivated soils annual
36 input of new Nr is approximately 9.7 Tg from fertilizer N, 1.1 Tg from livestock manure
37 (recycled N), ~7.7 Tg from biological N fixation and 1.2 Tg from atmospheric deposition.
38 Assuming that loss of fertilizer N from the small area of organic soils is a minor fraction of the
39 total, then ~17% of N input from synthetic fertilizer, ~12% of total N input, is stored in cropland
40 mineral soils annually.

Table 3-15. Net Annual Soil C and N Change: Cropland Remaining Cropland, 2002 (negative sign indicates a decrease in storage: positive number indicates increase in storage, soil C/N ratio = 12)(USEPA, 2007b)

	-----Tg-----	
	C	N
Mineral Soil	17.1	1.42*
Organic Soil	-8.3	-0.69
Net Annual Soil C and N Change: Land Converted to Cropland, 2002		
	0.8	0.067
Total	9.6	0.80

**In the light of recent publications that discuss possible overestimates of soil organic matter content conservation by no-till cropping practices, the above estimates of soil C and N storage in mineral soils may need additional consideration. (Baker et al. 2007; Blanco-Canqui, H. and R. Lal. 2008). These authors 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. also show that long-term, continuous gas exchange measurements have not detected C gain due to reduced tillage. They concluded that although there are other good reasons to use conservation tillage, 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.*

3.3.2.2. 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; (http://www.nrcs.usda.gov/technical/land/nri03/national_landuse.html)). The USEPA Inventory of U.S. Greenhouse Gas Emissions and Sinks 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 we use the NRCS value of 42.9 million ha 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 losing biomass and returning N to the environment. The USEPA (2007b) does not estimate carbon changes in turf grass, but do 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) (USEPA,

2007b). Annual fertilizer N input into the urban landscape is approximately 10% of total fertilizer N consumption in the U.S. (USEPA, 2007b), 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 N_r input annually.

3.3.2.3. *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 USEPA (2007b) 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 NW. Birdsey (1992) estimated that 52.5 X 10³ 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 USEPA estimate for 2002 is 43.6 X 10³ Tg of C. For the following N storage estimate, based on USEPA (2007b) 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 3-16)

Table 3-16. Forest Net Annual C and N Change: Forest and Harvested Wood Products, 2002 (negative sign indicates a decrease in N in storage: positive number net storage loss) (USEPA, 2007b).

	-----Tg-----	
	<u>C</u>	<u>N</u>
Above ground biomass	84.6	0.32
Belowground biomass	16.4	0.063
Dead Wood	9.1	0.035
Litter	7.2	0.028
Soil Organic Matter	-2.8	-0.23
Harvested Wood	58.5	0.22
Total	173	0.43

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 nitrogen status 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 carbon stocks were estimated using the Century biogeochemical model and data used were based upon the NRCS/NRI survey (USEPA, 2007b). Changes in soil N content were estimated using a C/N ratio = 12. Nitrogen input into rangelands is generally only from atmospheric deposition which

1 contributes 1.9 Tg N each year to range production (Table 3-17). Rangeland tends to be in
 2 relatively remote areas where atmospheric Nr deposition is low.
 3

4 **Table 3-17. Grassland net annual C and N change (negative sign indicates a decrease in**
 5 **storage: positive number indicates gain in storage, soil C/N ratio = 12)(USEPA, 2007b).**

6 Net Annual Soil C and N Change: Grasslands Remaining Grasslands, 2002

	Tg C	Tg N
8 Mineral Soil	-0.8	-0.067
9 Organic Soil	-1.3	-0.108

10
 11 Net Annual Soil C and N Change: Lands Converted to Grasslands

	5.8	0.48
--	-----	------

15 Total	3.7	0.31
----------	-----	------

17
 18 Collectively, forests and grasslands stored ~0.74 Tg of N in 2002. Much of the soil N storage in
 19 grasslands is a result of conversion of croplands to grasslands, mainly due to the conservation
 20 reserve program. Forest soils appear to be losing N while overall N storage is from
 21 accumulation in above ground biomass and that that remains in forest products that are stored for
 22 long periods.
 23

24 *3.3.2.4. Summary of estimates of Nr stored in terrestrial systems in 2002*

25
 26 An estimated 1.7 Tg of N was stored in the terrestrial systems of the contiguous 48 states in 2002
 27 (Table 3-18). Soils were the largest reservoir with croplands (0.82) and grasslands (0.31)
 28 sequestering most of the N. Estimated total Nr input from synthetic fertilizer, biological N
 29 fixation and atmospheric deposition into terrestrial systems within the contiguous 48 states in
 30 2002 was ~25 Tg . Although uncertainty of the storage estimate needs to be assessed, it is
 31 probably at least +/-50%. Annual storage in agricultural, grassland and forest soil and in forest
 32 biomass is approximately 6 to 10 % of annual Nr input. All of the input and outflow numbers
 33 are highly uncertain, but N loss through denitrification appears to be the major loss mechanism.
 34 As with the 16 NE USA watershed example, discussed in section 3.3.3.2, storage accounts for
 35 only a small (17 %) portion of the annual N input while apparent loss through denitrification
 36 dominates the budget. Some small fraction is re-volatilized and exported from the continent.
 37

38 **Table 3-18. Summary of estimates of Nr stored in terrestrial systems in 2002.**

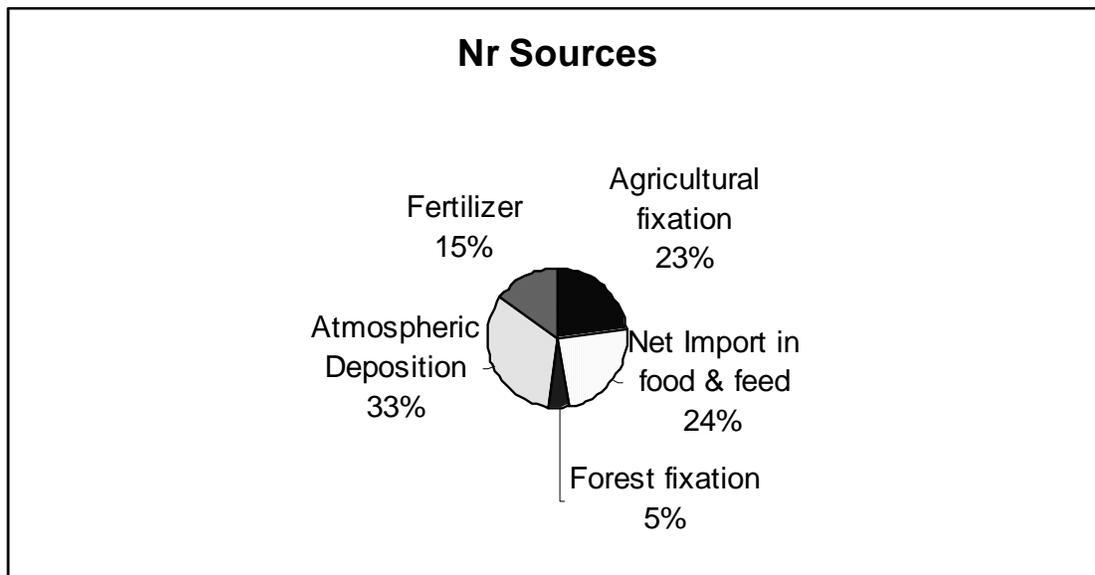
System	Estimated Tg N Stored yr ⁻¹
42 Agricultural	0.80
43 Populated	0.12
44 Vegetated	
45 Forest	0.43
46 Grassland	0.31
47 Total	1.66

1 **3.3.3. Flows of Nr in the USA for 2002: Estimated budget and flows of Nr for the entire**
2 **continental USA, and a detailed analysis of a Nr budget for 16 watersheds in the NE USA.**
3

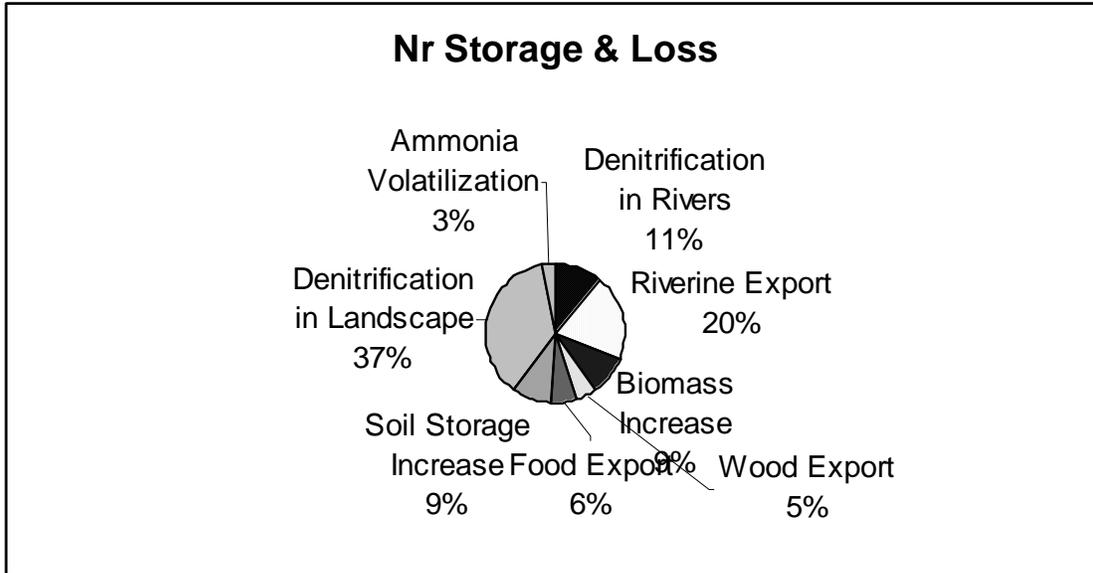
4 There are no comprehensive data available to assess the transfer and transformations in and
5 between the atmosphere, terrestrial systems (agriculture, populated and vegetated systems) and
6 aquatic systems nationally. **Determining a national N budget is a priority research area.** As
7 there are no national data available, an example watershed from the NE USA, for which data are
8 available, is used to show an evaluation of the inputs and fate of Nr (Fig. 3-14) (Van Breemen et
9 al. 2002). This analysis is followed by a “best guess” attempt is made to assign numbers to the
10 Nitrogen Cascade for the continental USA in 2002 (Fig. 3-15).

11
12 *3.3.3.1. Nr input and losses within a large watershed in NE USA.*
13

14 The watershed in this study included 16 catchments, which encompass a range of climatic
15 variability, Maine to Virginia. The watershed is a major drainages to the coast of the North
16 Atlantic Ocean. Using data from the early 1990s, Boyer et al. (2002) reported the quantification
17 of N inputs to each catchment from atmospheric deposition, nitrogen fertilizers, biological N
18 fixation, and import of N in agricultural products (food and feed). They compared inputs with N
19 losses from the system in riverine export. As a part of the same study, Van Breemen et al.
20 (2002) analyzed the fate of nitrogen inputs to these watersheds and developed budgets for each
21 watershed. The total area of the watersheds was 32,666 km² with land use categories of forest
22 (72%), agricultural (19%), urban (3%), wetlands (5%), and 1% other uses. The Nr input into the
23 watersheds (using weighted averages for all 16 watersheds) was 3420 kg km⁻² yr⁻¹. Figure 3-15
24 shows the Nr sources and the estimated fate of this Nr as a per cent of the weighted average Nr
25 input.
26
27



28
29



1
2 **Figure 3-14. Nr input and loss from 16 catchments of a large watershed in the NE USA. The**
3 **Nr input into the watersheds (using weighted averages for all 16 watersheds) was 3420 kg km²**
4 **yr⁻¹ (Van Breemen et al. 2002).**

5
6 Van Breemen et al. (2002) indicate that Nr inputs and storages and losses were well correlated
7 ($R^2 = 0.98$). Denitrification in landscape soils is the most uncertain estimate, because rates are
8 calculated by difference between total inputs and outputs, so they accumulate errors from all
9 estimates. They suggest that the denitrification loss term may also reflect the change in N
10 storage in groundwater. The net storage of N in the soil (18% of total storage and losses)
11 indicates that there is a non-steady state condition in the soil. Increasing storage of Nr on land
12 implies that drainage and denitrification exports of Nr are likely to increase when a new steady
13 state condition is reached.

14
15 These data suggest that Nr research need to focus on understanding the “denitrification” loss
16 term in this analysis. The losses occur in the terrestrial landscape, before Nr enters the river.
17 Where do these losses occur, within the agricultural field, in drains and ditches near the
18 agricultural field, in riparian areas, or wetlands? Understanding this term may help in the
19 management of Nr in watersheds to decrease nitrate movement into aquatic systems as well as to
20 limit N₂O emissions to the atmosphere.

21
22 The Van Breeman et al. (2002) study also estimated that approximately 30% of N input was
23 exported to the rivers and about two thirds (20% of total N input) of this N was exported to
24 coastal waters by rivers. The remaining one third (11% of total N input) was considered to have
25 been denitrified in the rivers. These examples also demonstrate that Nr in the atmosphere,
26 terrestrial systems and aquatic systems are not separate and must be considered collectively.
27 Atmospheric deposition is a variable, but important input into aquatic systems that contributes to
28 Nr enrichment problems. Aquatic and terrestrial systems process this Nr and return other Nr
29 gases (NH₃, NO_x and N₂O to the atmosphere). Nr from terrestrial systems impacts both the
30 atmosphere and aquatic systems through emission of ammonia, NO_x, N₂O and leaching and
31 runoff of nitrate.

3.3.3.2. *Estimated budget and flows of Nr in the USA for 2002: An attempt to apply national numbers to the N Cascade Diagram*

On the national scale the flows of Nr can be divided, on annual basis, into Nr that is newly fixed and Nr that is recycled (Table 3-18). Fixation or mobilization of new Nr comes mainly from the combustion of fossil fuels, biological N fixation in crop and forage legumes, and synthetic fertilizer production. The following discussion uses very rough estimates (best guesses) of Nr inputs and flows for the USA in 2002. Data used were obtained mainly from FAO (2008), USEPA (2007b) and Table 3-1.

Table 3-19 Estimates of new and recycled in Nr in the USA in 2002 (Tg N).

	Tg N	
<u>New Nr</u>		
Combustion NOx & NHx	5.4	
Crop and Pasture N Fixation		14.1
Synthetic Fertilizers		10.9
Industrial Feedstock		4.2
Commodities Import		0.2
Total		34.8
<u>Annual Recycling</u>		
Agricultural NOx & NHx	1.9	
Livestock Manure		6.0
Human Waste		1.3
Crop Residue		4.4
Soil Organic Matter		4.7
Crop N (Total Production – Export)	5.3	
Meat/eggs/milk (Total – Export)	1.7	
Total		25.3

Table 3-20. Storage within and transfers of Nr within and between Environmental Systems.

Agricultural		
Soil to crop		4.7
Soil to livestock*	7.6	
Livestock to soil	1.2	
Agricultural to Atmosphere		
Agricultural to Urban (food)		2.0
Populated to Aquatic (human waste)		1.3
Terrestrial to Atmospheric		
Terrestrial + Aquatic N ₂ O	0.8	

Terrestrial Storage 1.7

*N from forage production that was derived from soil N mineralization

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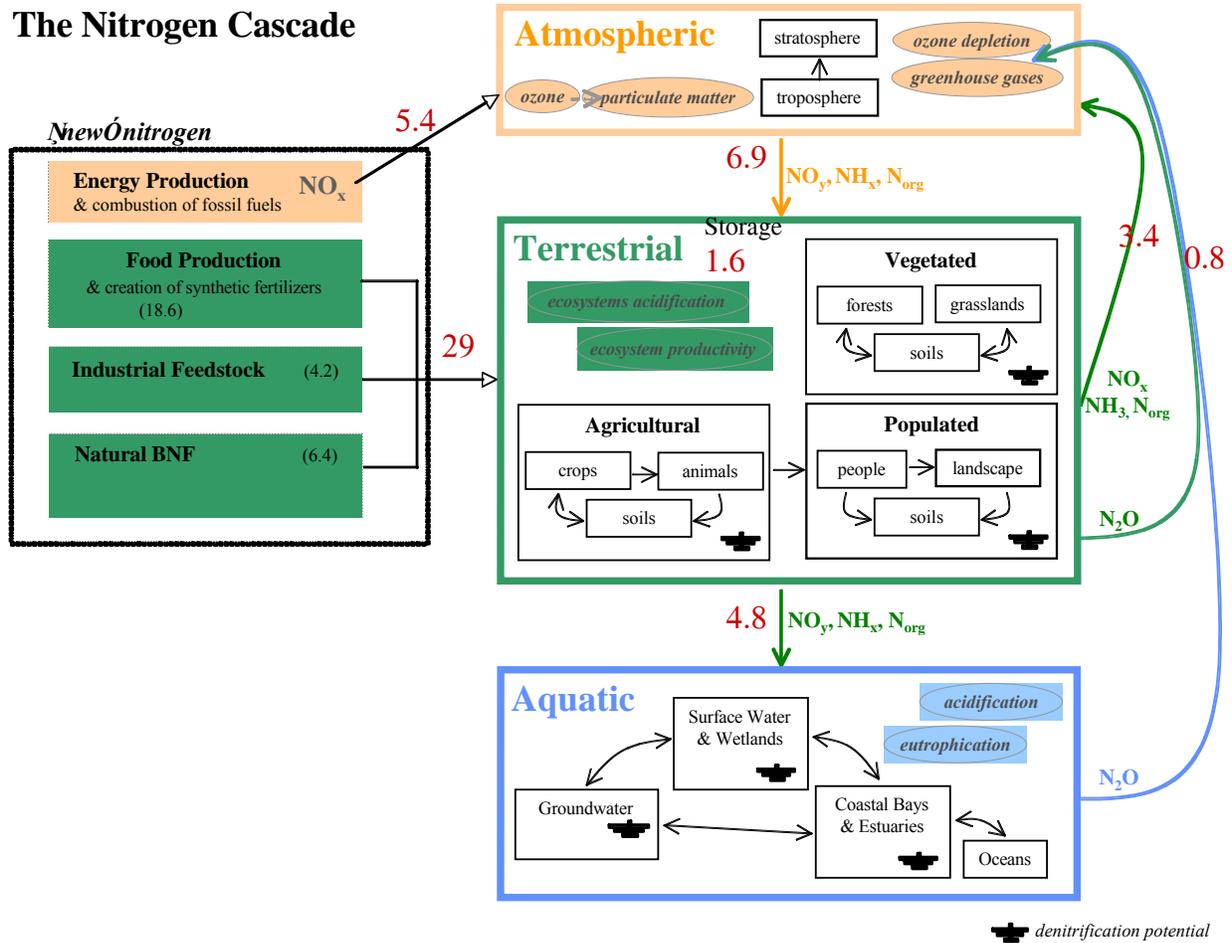
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1 The application of the above numbers (Tables 3.1, 3-18, 3-19 and 3-20) to the full Nitrogen
2 Cascade provides the estimated input, storage, and flow of Nr between the atmospheric,
3 terrestrial and aquatic environmental systems (Fig. 3-16). Fossil fuel combustion inputs ~6 Tg of
4 Nr, mostly in the form of NO_x, to the atmosphere in while agriculture provides the majority of
5 another 3.1 Tg Nr to the atmosphere, mainly as NH_x. Nr input into terrestrial systems is derived
6 from synthetic fertilizer (10.9), industrial feedstock (4.2) atmospheric deposition (5.9) and
7 biological N fixation (7.7 from crops and 6.4 in unmanaged grasslands). Approximately 1.7 g of
8 N is stored annually in terrestrial systems in soils and forests. Terrestrial and aquatic systems
9 emit ~0.8 Tg of N₂O-N to the atmosphere annually as a result of nitrification and denitrification.
10 Outflow of Nr from terrestrial to aquatic systems was ~4.8 Tg Nr. Approximately 1.3 Tg of this
11 N came from sewage treatment facilities with the remainder mainly from leaching of nitrate from
12 soils.

13
14 **Finding F3-12:** Denitrification of Nr in terrestrial and aquatic systems is one of the most
15 uncertain parts of the nitrogen cycle. Denitrification is generally considered to be a dominant N
16 loss pathway in both terrestrial and aquatic systems, but it is poorly quantified

17
18 **Recommendation R3-18:** USEPA, USDA, DOE, and universities should work together to
19 ensure that denitrification in soils and aquatic systems is properly quantified, by funding
20 appropriate research.

The Nitrogen Cascade



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Figure 3-15. Estimated N budget and flows for the continental USA in 2002, Tg N.

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. We refer 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.

1 3.3.4.1. *Biophysical controls in Terrestrial Environmental Systems*

2
3 Approximately 36 Tg of new Nr is introduced into the U.S. each year (Table 3-1). This new Nr
4 is derived from consumption of ~11 Tg of synthetic nitrogen fertilizer, ~8 Tg of N is fixed by
5 biologically by crops, and ~ 5 Tg is emitted from fossil fuel combustion annually. This N is used
6 to produce food and fiber (~15 Tg) or is formed during electrical generation, industrial
7 production or transportation. Efforts to decrease the creation of new Nr should first look to
8 conservation.

9
10 Conservation of fossil fuel and resulting decrease in use of fertilizer in food and fiber production
11 or decreased Nr emission can come through a variety of mechanisms such as more energy
12 efficient industrial processes, energy efficient homes and vehicles. Further gains are possible
13 through conservation practices and alternatives to wasteful approaches, such as improving public
14 transportation to minimize use of personal automobiles, and to use local products that don't
15 require long-distance shipping.

16
17 Conservation in food and fiber production and food consumption can also play an important role
18 in limiting Nr. As agriculture is the largest consumer and producer of Nr, consumption of
19 fertilizer N could be decreased by changes in diet and increasing fertilizer N use efficiency in
20 crop and fiber production systems. The "choke" points discussed in this section include: protein
21 consumption in the human diet, removing croplands that are susceptible to Nr loss from crop
22 production, decreasing fertilizer N demand by increasing fertilizer use efficiency in crop and
23 fiber production, turf grass and nitrogen fertilization in the USA, and managing Nr during
24 recycling through livestock production.

25
26 *Decreasing the amount of fertilizer N needed through changes in human diet.* Along with
27 increasing fertilizer N use, continued high intake of protein in developed countries and changes
28 in the diet of people in developing countries will likely lead to greater N losses from global food
29 production in the future. The first aspect of changes in food production concerns increasing
30 protein consumption as global population increases and gets wealthier, which is likely to require
31 increased N input into food production (Galloway et al. 2007; Naylor et al, 2005).

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

44
45 The reason for focusing on the consumption of animal protein is that more N is needed to
46 produce a unit of animal protein than an equal amount of grain protein. Bleken et al. (2005) note

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1 that the N cost of animal production in Norway and the Netherlands was approximately five
2 units of N in feeds for each unit of N produced. Approximately 2.5 units of N are required to
3 produce a unit of wheat protein-N. Bequette et al. (2003) report that dairy cattle consume four
4 units of N in feeds (including forage and grains) for every unit of N that appears in milk. Using
5 a range of efficiencies for animal production practices, Kohn et al. (1997) estimated that 4 to 11
6 units of fertilizer N would be used in a whole farm system to produce a unit of milk protein.

7 This ratio would be lower when using legume N to feed cattle, as is commonly done. Based
8 upon the extra N required to produce animal protein compared to grains, continued high protein
9 consumption in developed countries and changes to higher protein diets in developing countries
10 will likely increase N input and losses in food production.

11
12 Moderating this increase by decreasing the average amount of total protein consumed in
13 developed countries is one mechanism of limiting part of the expected increased N requirement
14 in food production. One example of a country with a good food supply and moderate
15 consumption of animal protein is Italy in 1963. At that time food supply was adequate to ensure
16 sufficient nutrition to all groups of society (Bleken 1997). Total protein consumption was 85 g
17 per capita d⁻¹, and consumption of animal protein was 32 g, roughly half of the current U.S. diet,
18 and yet much higher than the average of developing countries. Another example is Japan, where
19 animal protein consumption has traditionally been low, although it has increased from 25 g in
20 1963 to 54 g animal protein per capita d⁻¹ in 1995. In the same period the total protein
21 consumption has increased from 73 g to 96 per capita d⁻¹.

22
23 Bleken (1997) analyzed the relation between human diet and global N need for food production.
24 Her analysis indicates that the total N needed for diets with high animal protein intake
25 (comparable to many industrialized countries today) are almost twice as high as the N needed for
26 the average diet in Italy 1963, mentioned above, or for Turkey in 1993. Based on her analysis,
27 we assume that in the high-N input regions per capita N need for food production may be
28 reduced by 45%, which would reduce present-day N inputs by 15% worldwide.

29
30 Switching to a lower protein diet may not, however, reduce N losses if the new diet includes
31 increased quantities of fruits, vegetables, and nuts, in addition to staple grains, beans and pulses.
32 Vegetables, fruit and nuts are high value crops that typically require large inputs of fertilizers and
33 pesticides when produced at a large, commercial scale, and nitrogen fertilizer losses can be
34 considerably larger than for grain crops. Having a very diverse diet that includes a wide range of
35 high value fruits and vegetables that are available 365 days a year whether they are in-season
36 locally or not, also have consequences for N inputs/outputs from agriculture--both within the
37 USA and globally. Additional N_r may be conserved by decreasing the amount of food that is
38 wasted.

39

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1 *Removing croplands that are susceptible to Nr loss from crop production.* Booth and Campbell
2 (2007)'s model analysis of nitrate loading in the Mississippi River Basin provides estimates of N
3 input from agricultural lands to be similar to those estimated by Del Grosso et al. (2006). These
4 recommendations are essentially the same as those arrived at in the original national hypoxia
5 assessment which suggested that the most leaky lands be taken out of production (Doering et al.
6 1999). Booth and Campbell state that,

7
8 *“Nitrogen derived from fertilizer runoff in the Mississippi River Basin (MRB) is*
9 *acknowledged as a primary cause of hypoxia in the Gulf of Mexico. To identify the*
10 *location and magnitude of nitrate runoff hotspots, and thus determine where increased*
11 *conservation efforts may best improve water quality, we modeled the relationship*
12 *between nitrogen inputs and spring nitrate loading in watersheds of the MRB. Fertilizer*
13 *runoff was found to account for 59% of loading, atmospheric nitrate deposition for 17%,*
14 *animal waste for 13%, and municipal waste for 11%. A nonlinear relationship between*
15 *nitrate flux and fertilizer N inputs leads the model to identify a small but intensively*
16 *cropped portion of the MRB as responsible for most agricultural nitrate runoff.*
17 *Watersheds of the MRB with the highest rates of fertilizer runoff had the lowest amount*
18 *of land enrolled in federal conservation programs. Our analysis suggests that scaling*
19 *conservation effort in proportion to fertilizer use intensity could reduce agricultural*
20 *nitrogen inputs to the Gulf of Mexico, and that the cost of doing so would be well within*
21 *historic levels of federal funding for agriculture. Under this simple scenario, land*
22 *enrolled in conservation programs would be increased by about 2.71 million hectares, a*
23 *29% increase over 2003 enrollments, while land taken out of traditional fertilized*
24 *agriculture and enrolled in conservation programs would constitute about 3% of 2003*
25 *fertilized hectares.*

26 *Would increasing conservation to this extent be sufficient to reduce nitrate*
27 *loading and the size of the Gulf hypoxic zone? It has been estimated a 30% reduction in*
28 *total nitrogen inputs would shrink the hypoxic zone by 20-60%, though still greater*
29 *reductions may be required. Our model characterizes nitrate loading, not total nitrogen*
30 *loading, but under our scenario, a 30% reduction in total nitrate inputs to the Gulf would*
31 *require a 50% reduction in agricultural loading to aquatic systems.”*

32
33 The latest Gulf of Mexico hypoxia report indicates that an even greater Nr reduction is needed to
34 get appreciable decrease of the hypoxic zone (USEPA, 2007a).

35
36 *Decreasing fertilizer N demand by increasing fertilizer use efficiency in crop and fiber*
37 *production.* The largest input of reactive nitrogen (Nr) in North America is nitrogen (N)
38 fertilizer used for crop production. The mean annual N fertilizer input to North America
39 between 1999 and 2003 was 12.5 Tg (million metric tons). Of this fertilizer N, 66% was used to
40 fertilize cereal crops, mainly corn and wheat (Dobermann and Cassman, 2005).

41
42 Corn yield in the USA has increased from an average of 100 bu/ac in 1985 to 136 bu/ac in 2005
43 as a result of improved nutrient and pest management, expansion of irrigated area, conservation
44 tillage, soil testing, and improved crop genetics (yield and pest resistance) (CAST, 2006). From
45 1980 to 2000, N-fertilizer use efficiency (NFUE, kg grain produced per kg applied N) increased
46 from 42 to 57 kg kg⁻¹, a 35% efficiency gain during a period when average USA corn yields
47 increased by 40% (Fixen and West, 2002). Despite this steady increase in NFUE, the average N
48 fertilizer uptake efficiency for corn in the north-central USA was 37% of applied N in 2000

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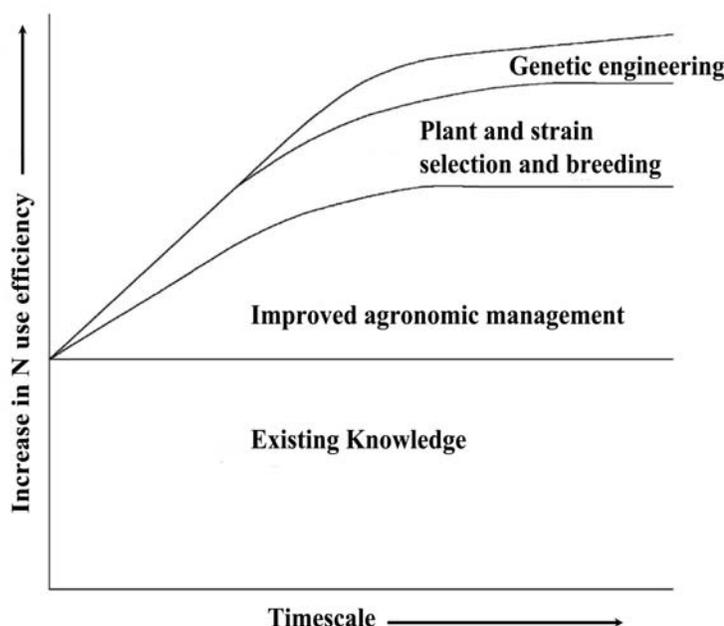
1 based on direct field measurements (Cassman et al. 2002). These results indicate that a large
2 majority of the applied N fertilizer is vulnerable to loss pathways such as volatilization,
3 denitrification, runoff, and leaching. The results also suggest there is substantial room for
4 improvement in N efficiency currently achieved by farmers.

5
6 Although progress has been made to increase both cereal yield and NFUE, a concerted effort to
7 further increase NFUE remains a logical control point to reduce production costs, because N
8 fertilizer represents a significant input cost, and to limit Nr leakage (e.g. NH₃, NO_x, N₂O, NO₃⁻)
9 from agroecosystems.

10
11 The goal of reducing Nr while sustaining adequate rates of gain in cereal production to meet
12 expected food demand will require increases in both total nitrogen use efficiency (NUE) and
13 NFUE, which in turn will require innovative crop- and soil management practices. This need is
14 exacerbated by the recent increase in demand for corn to produce ethanol biofuel. The concept of
15 improved N synchrony—practices that better match the amount, timing, and geospatial location
16 of applied N to crop-N demand and the N supply from indigenous soil resources-- is generally
17 viewed as the most appropriate approach for improving NUE (e.g. Appel, 1994; Cassman et al.
18 2002). The challenge is for greater synchrony between crop N demand and the N supply from
19 *all sources* (e.g. soil, fertilizer, organic inputs such as manure, compost, or green manures, etc)
20 throughout the growing season. Losses from all N-loss mechanisms increase in proportion to the
21 amount of available N present in the soil profile at any given time.

22
23 Several promising technologies and combinations of technologies have emerged in recent years.
24 Significant increases in NUE are often achieved through reductions in N fertilizer use by 10 to
25 30 %, while increases in yield tend to be small (Giller et al. 2004). Figure 3-17 indicates where
26 expected greatest gains in NUE are to be realized in the future from different technology options.
27 Each of these improvements in management and genetics helps to better match the amount and
28 timing of applied N to crop-N demand and the N supply from indigenous resources. However,
29 large investments in research, extension education, and technology transfer will be required to
30 achieve the degree of improved synchrony needed to make substantial improvements in NUE.
31 The need to accelerate the rate of gain in crop yields to meet increasing demand for human food,
32 livestock feed, and biofuels represents an additional new challenge. Crop prices are expected to
33 rise as they more closely track the price of petroleum (CAST, 2006). Higher crop prices will
34 motivate farmers to achieve higher yields, and higher crop yields require a greater amount of N
35 uptake to support increased biomass production (Greenwood et al., 1990). Therefore, an explicit
36 emphasis on developing technologies that contribute to both increasing yields and NUE will be
37 needed to ensure that the goals of food security, biofuel production, and protection of
38 environmental quality are met.

39



1
2 **Figure 3-16.** *The likely impact of research investment in increasing N Use Efficiency (Giller et al.*
3 *2004).*

4
5 **Finding F3-13:** Current production of biofuels uses grain and in the future may use forages that
6 would otherwise be available for animal feeds and human foods. Byproducts from fuel
7 production are high in protein and minerals and may decrease the need for high-protein crops to
8 feed livestock. Increased biofuel production may increase Nr flow from agricultural systems.

9
10 **Recommendation R3-19:** Ensure that USEPA, USDA, DOE, in collaboration with other
11 Federal and State agencies, provide appropriate life cycle analyses on which policy decisions for
12 biofuel production and use of co-products are based.

13
14 *Managing Nr during recycling through livestock production.* Newly fixed Nr is produced
15 biologically or added as fertilizer to meet the demand for food and fiber production. Much of the
16 N is used in cereal crop production and cereal crops are then used to feed livestock. The new Nr
17 is then recycled through the livestock production system and becomes again susceptible to losses
18 to the atmosphere as ammonia and NO_x, is available for additional N₂O production, and
19 movement into aquatic systems as ammonium and nitrate.

20
21 The bulk of the N fed to livestock ends up in manure, and where this manure (~ one half in urine
22 and one half in feces) is produced, there is often a much greater supply than can be efficient or
23 economically used as fertilizer on crops. For large animal feeding operations (AFO's) there is
24 considerable expense associated with disposal of the manure. Various storage systems have been
25 developed to deal with this excess manure, the most interesting of which, from the standpoint of
26 integrated policy on N, convert the urea to N₂. These represent a choke point where reactive N is
27 removed, on time scales of millennia, from biogeochemical cycles. The fraction of the feed N

1 that is converted to N₂ or even can be converted to N₂ remain major unanswered scientific or
2 technical questions; this brief report reviews the current state of knowledge.

3
4 The NRC (2003) report bemoaned the paucity of credible data on the effects of mitigation
5 technology on rates and fates of air emissions from AFO's, but called for their immediate
6 implementation. That report also called for a mass balance approach in which the losses of N
7 species such as NH₃, NO, N₂, and N₂O are expressed as a fraction of the total N loss. Quoting
8 from the NRC report:

9
10 *Storage covers for slurry storage tanks, anaerobic lagoons, and earthen*
11 *slurry pits are being studied as a method to decrease emissions from those*
12 *containments. Both permeable and impermeable covers are being studied.*
13 *Tested covers range from inexpensive material such as chopped straw (on slurry*
14 *containments only) to more expensive materials such as high density*
15 *polyethylene. Covers can decrease emissions from storage but their net effect on*
16 *emissions from the system is conditional on how the effluent is used on the farm.*

17 *Anaerobic digestion in closed containment has been studied for many*
18 *types of applications. Anaerobic digestion is the process that occurs in an*
19 *anaerobic lagoon. When conducted in closed vessels, gaseous emissions*
20 *including methane, carbon dioxide and small amounts of other gasses (possibly*
21 *ammonia, hydrogen sulfide, and VOCs) are captured and can be burned for*
22 *electricity generation, water heating, or simply flared. The in-ground digester*
23 *being tested on a swine farm in North Carolina is an example of the ambient*
24 *temperature version of this technology (there are also mesophilic and*
25 *thermophilic designs). The concentration of ammonia remaining in effluent from*
26 *that digester is higher than the concentration in lagoon effluent and can be*
27 *volatilized once exposed to air.*

28
29 Recent research (e.g., Shores et al. 2005; Bicudo et al. 2004; Funk et al. 2004a; Funk et al.
30 2004b) demonstrates reduction in NH₃ emissions after a permeable cover was installed. Miner
31 et al. (2003) reported that a polyethylene cover can reduce NH₃ emissions by ~80%, but it is not
32 clear what fraction of that N was converted to N₂. Harper et al. (2000) reported that in a well-
33 managed swine lagoon denitrification N₂ losses can be equivalent to N lost as ammonia, in other
34 words about 50% efficiency. Kermarrec et al. (1998) reported that sawdust litter helps reduce
35 NH₃ emissions from pig manure with 44-74% of manure N converted to N₂, but > 10% of the
36 manure N was released as N₂O. Sommer (1997) cattle and pig slurry tanks NH₃ 3.3 kg N m⁻² yr⁻¹
37 until covered with straw then below detection limit. Mahimairaja et al. (1994) reported that
38 NH₃ volatilization was reduced by 90-95% under anaerobic conditions. See section 3.2 for a
39 discussion of best management practices to minimize NH₃ emissions from livestock waste.

40
41 **Finding F3-14:** Enhanced denitrification in manure handling has the potential to reduce NH₃
42 emissions substantially, but the amount of N₂O produced during the process is uncertain and the
43 denitrification potential number cannot be quantified to better than a factor of two.

44
45 **Recommendation R3-20.** Continue to research the rate and extent of denitrification in manure
46 handling facilities.

1
2 **Recommendation R3-21.** EPA should work with USDA to encourage implementation of best
3 practices at the farm and AFO level.

4
5 *Wetlands to decrease nitrate loading of aquatic systems.* The 40 to 60% of fertilizer N that is
6 not used by crop production, and an appreciable portion of the N fixed by soybeans, is moved
7 from the crop field into surface and groundwater. Del Grosso et al. (2006) estimate that
8 nationally, 30+% as much as the N applied as fertilizer is susceptible to leaching. In soybean
9 production, where little fertilizer N is used, nitrate leaching still poses a significant problem. Del
10 Grosso et al. (2006) estimate that 93% as much nitrate is leached under soybean production as
11 under corn production. Much of the nitrate leached from agricultural fields could be removed
12 from drainage water in wetlands, either natural or reconstructed.

13
14 Nitrate removal from the water column in wetlands is performed by plant uptake, sequestration
15 in the soils, and microbial transformation that include immobilization and denitrification. Plant
16 uptake and microbiological immobilization result in temporary storages in the system since most
17 nitrogen will eventually return to the wetland via plant death and decomposition. In contrast,
18 denitrification constitutes a real nitrogen sink because in this process bacteria reduce nitrate to
19 nitrogenous gases (N₂, NO, N₂O) that are emitted to the atmosphere (Clement et al., 2002). In
20 general, nitrate removal by wetlands, primarily caused by microbial denitrification, varies
21 seasonally, with highest rates during summer and lowest rates during the coldest temperatures
22 (Mitsch et al., 2000; Spieles and Mitsch, 2000; Hernandez and Mitsch, 2007). Hernandez and
23 Mitsch (2007) found that permanently flooded wetlands had lower N₂O/N₂ ratios of emissions
24 than did intermittently flooded wetlands. They also found that the ratio was higher in the cold
25 months even though the flux rates are much lower then. A full risk assessment needs to be made
26 to determine how much pollutant swapping, i.e., exchanging nitrate for N₂O is advisable.

27
28 In addition to preserving existing wetlands there are two basic approaches utilizing wetlands for
29 reducing the nitrogen and other nutrients from reaching rivers and streams and especially
30 vulnerable downstream coastal systems: 1) creation and restoration of ecosystems, principally
31 wetlands and riparian forests, between farms and adjacent ditches, streams and rivers; and 2)
32 diversion of rivers into adjacent constructed and restored wetlands all along the river courses
33 (See Chapter 4).

34
35 At a series of workshops on restoration of the MOM (Mississippi-Ohio-Missouri) River Basin in
36 2003-04 (Day et al., 2005; Mitsch and Day, 2006), scientists and managers were asked to focus
37 on needed research and chokepoint opportunities, especially for managing nitrogen in that basin.
38 They concluded that a major, interdisciplinary research program, as a lead-in to the actual
39 restoration of wetlands and rivers, needs to take place with sufficient funding, study sites, and
40 time to reduce remaining uncertainties about the efficacy of wetlands to solve pollution problems
41 related to nitrogen. Twenty to thirty full-scale, existing and new agricultural/wetland
42 demonstration projects should be located throughout the country and instrumented to study
43 agricultural runoff into wetlands in a variety of soil conditions. Pilot and full-scale studies are
44 needed of diversions into riparian systems along river channels to determine their effectiveness
45 for nutrient removal.

1 To give scale the solution needed, restoration of over 2 million hectare of wetlands is needed in
2 the MOM basin to reduce the nitrogen load to the Gulf of Mexico sufficiently to ensure a
3 reduction in the size of the hypoxia (Mitsch et al., 2001; Mitsch and Day, 2006; see Chapter 4).
4 If wetlands could be economically and effectively restored where croplands now exist on hydric
5 soils within the 100-year floodplain, returning croplands that are on hydric soils may be an
6 important nitrate control mechanism. Cropland on hydric soil in the floodplain occupy about 2.8
7 million hectare, 40% more than is needed for the restoration. If this area and its wetlands were
8 given back to the Mississippi, over a million tons of nitrate-nitrogen would be annually removed
9 or prevented from reaching the Gulf of Mexico (Hey et al. 2004).

10 Given the interactions among oxidized and reduced nitrogen species, it is important to recognize
11 the potential for unintended consequences to occur as a result of strategies aimed at limiting one
12 form of Nr in air or water that can lead to the increased production of other forms of Nr. One
13 such instance is the potential offsetting of the benefits of nitrate remediation at the expense of
14 increasing input of nitrous oxide to the atmosphere (See section 3.4.7.3).

15
16 *3.3.4.2. Technical controls (control points) on transfer and transformations of Nr in and*
17 *between environmental systems: Nitrogen Oxides*

18
19 A major contributor to Nr in the atmosphere is fossil fuel combustion. During the combustion
20 process nitrogen oxides ($\text{NO}_x = \text{NO} + \text{NO}_2$) are released to the atmosphere. Globally the
21 production of NO_x has accelerated the last few decades through, primarily the increase in fossil
22 fuel combustion (Galloway et al., 1995; 2008). With this increase in emissions from ~5 Tg N in
23 1940 to ~ 25 Tg N in 2005, combustion of fossil fuels account for about 50% of the total global
24 NO_x emissions for 1990. Of the anthropogenic sources, fossil fuel, aircraft, biomass burning,
25 and part of the soil emission are most important (Holland et al., 1997). Although global NO_x
26 emissions continue to increase, these emissions are declining in the U.S. (see section 3.3.1).

27
28 Nitrogen oxide is formed during combustion by three mechanisms:

- 29
- 30 • thermal NO_x where nitrogen and oxygen gas, present normally in combustion air,
31 combine at high temperatures, usually above 1600 C to form NO through the Zeldovich
32 mechanism.
33
 - 34 • fuel NO_x where nitrogen from a fuel, e.g., coal and biofuels, is released as some
35 intermediate and then combines with oxygen to form NO, and
36
 - 37 • prompt NO_x where nitrogen gas combines with radical components of the fuel, forming
38 various compounds including hydrogen cyanide and other cyano radicals. These in turn
39 form NO_x . Contributions of prompt NO_x are usually low as compared to fuel NO_x .
40

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

44 Reduction of the temperature limits the kinetics of the nitrogen/oxygen reaction. Temperature
45 can be controlled by using a fuel-rich mixture versus fuel lean. In this case the reactions to take

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1 place at lower temperatures. Fuel-rich mixtures also reduce the amount of oxygen available for
2 reaction and there are changes to the chemical mechanisms which limit the oxidation of N₂. If
3 fuel lean mixtures are used for temperature control, while the temperature is lower, there is a
4 significant amount of O₂ present. Typically in external combustion systems, this is implemented
5 by using less excess air and using staged combustion. In addition, flue-gas recirculation (FGR)
6 is used to lower the temperature. Low-NO_x burners operate under the principle of internally
7 staging the combustion. To reduce fuel NO_x, air and fuel staging are used to reduce the peak
8 temperature where air and fuel are admitted in separate locations.

9
10 Chemical reduction of NO_x is also possible. These methods include: Selective Non-Catalytic
11 Reduction (SNCR); Selective Catalytic Reduction (SCR); and fuel reburning. SNCR is an add-
12 on technology where urea or ammonia is injected in a controlled temperature zone to allow for
13 the reduction of NO_x. SCR is also an add-on technology where the flue gas must pass through a
14 catalyst bed to allow for reaction between ammonia and NO_x. Care must be taken with both
15 technologies to avoid ammonia slip. Fuel reburning requires the injection of a fuel to create a
16 zone where NO_x is reduced to N₂. Low NO_x burners may also use an internal fuel reburning to
17 reduce the NO_x.

18
19 For internal combustion engines, the same mechanisms as discussed above are used but in a
20 variety of different ways, since these systems are using high pressure and predominately have
21 thermal NO_x versus fuel NO_x formation. Most technologies involve the need to reduce the peak
22 temperature and duration of high temperatures of the combustion zone. For example, gas
23 turbines utilize low NO_x burners, while spark ignition engines utilize a three-way catalyst which
24 requires less than 0.5% oxygen. In this case, additional NO_x is reduced by utilizing unburned
25 fuel as a reagent over the catalyst for chemical reduction of NO_x. It should be noted however,
26 that a side reaction for the three-way catalyst system is ammonia. For diesel engines, delaying
27 the injection of the fuel, and for spark ignited engines retarding the timing can reduce NO_x
28 emissions. Engines also use exhaust gas recirculation (EGR) to reduce the peak temperatures.
29 Recent road side studies have indicated high efficiency (~90%) for NO_x removal from the
30 American light-duty fleet [*Bishop and Stedman, 2008*].

31
32 **Finding F3-15:** Emissions of reactive nitrogen from fuel combustion (in the form of NO_x) have
33 been reduced substantially for some classes of emitters such as power plants and light duty
34 vehicles. Other sources, including most off road vehicles, some industrial equipment, and some
35 older electricity generating units, operate with little or no NO_x controls. Most sources can be
36 controlled (with well established engineering practices and at a reasonable cost) to the point of
37 90% reductions of NO_x emissions relative to uncontrolled combustion. NO_y levels in the
38 atmosphere remain too high to protect public health and welfare, and continued reductions of
39 NO_x emissions are necessary.

40
41 **Recommendation R3-22.** Decrease NO_x emissions from off road vehicles, some industrial
42 equipment, and some older electricity generating units that currently operate with little or no
43 NO_x controls.

44
45 **Recommendation R3-23.** Improve monitoring and modeling of NO_x from vehicles and other
46 mobile sources.

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3.3.5. Critical numbers on budgets and flows that are either highly uncertain and/or are important for EPA

In this assessment of Nr transfers and transformations in and between the environmental systems of the N cascade we have encountered a number of areas where quantities or flows of Nr are highly uncertain. The areas important for EPA and in need of further quantification include:

1. Total denitrification in animal feeding operations, in soils, and in aquatic systems; all gaseous products that are produced and released to the atmosphere during denitrification, NO_x, N₂O and N₂.
2. The amount of Nr deposited in each environmental system as dry deposition needs to be quantified and monitored.
3. The amount of N₂O released during nitrification/denitrification in animal feeding operations, soils and aquatic systems.
4. NO_x exchange between soils and the atmosphere in terrestrial systems needs to be assessed.
5. Rates and amount of ammonia emission from fertilized soils and animal feeding operations.
6. The annual change in N storage in soils (agricultural, forest and grassland).

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1 **Appendix 3.3.**

2
3 **A. Production of N₂ and N₂O via gas-phase reactions**

4
5 Atmospheric conversion of NO_x and NH_x to less reactive N₂ or N₂O appears to play a minor role
6 in the global N budget, but currently is not well quantified. The gas-phase reactions in the
7 troposphere that convert NH₃ and NO_x to N₂ and N₂O, start with attack of NH₃ by OH:



10
11 Several potentially interesting fates await the NH₂ radical:



16
$$k_{\text{O}_3} = 1.9 \times 10^{-13} \text{ cm}^3 \text{ s}^{-1}$$

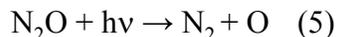
17
$$k_{\text{NO}_2} = 1.8 \times 10^{-11} \text{ cm}^3 \text{ s}^{-1}$$

18
$$k_{\text{NO}} = 1.8 \times 10^{-11} \text{ cm}^3 \text{ s}^{-1}$$

19
20 The first step, attack by OH, is slow. The rate constant for the Reaction 1 is $1.6 \times 10^{-13} \text{ cm}^3 \text{ s}^{-1}$
21 and the lifetime of NH₃ for a typical concentration of 10^6 OH cm^{-3} is about 70 d. In most areas
22 of the world where concentrations of NH₃ are high, concentrations of sulfates are also high, and
23 NH₃ is removed by conversion to condensed phase ammonium sulfate or bisulfate on time scales
24 much faster than 70 d. The mean lifetime of these aerosols with respect to wet deposition is
25 about 10 d.

26
27 There are some areas of the world, notably California and South Asia, where NH₃ and NO_x are
28 emitted in large quantities, but SO₂ is not, and there gas-phase conversion can take place. In
29 general, $[\text{O}_3] \gg [\text{NO}_x]$, and Reaction 2 represents an unimportant source of NO_x, but Reactions
30 3 and 4 may be atmospherically noteworthy. As an upper limit to current N₂O production, we
31 can assume that each of these regions covers an area of 10^6 km^2 and that they contain ammonia
32 at a concentration of $10 \mu\text{g N m}^{-3}$ in a layer 1000 m deep. The annual production of N₂ and/or
33 N₂O would then be on the order of 0.1 Tg N, a minor but nontrivial contribution to
34 denitrification and about 1% of the anthropogenic N₂O production. If NH₃-rich air is lofted out
35 of the boundary layer into the upper troposphere where deposition is impeded, it will have an
36 atmospheric residence time on the order of months, and the probability of reaction to form N₂O
37 or N₂ becomes greater. This possibility has not been investigated extensively. It is also possible
38 than Europe and North America will continue to reduce S emissions without reducing NH₃
39 emissions and the atmospheric source of N₂O will grow in importance.

40
41 In the stratosphere, N₂O photolysis leads to loss of Nr via



44
45 While reaction with an electronically excited oxygen atom O(¹D) leads to production of NO via



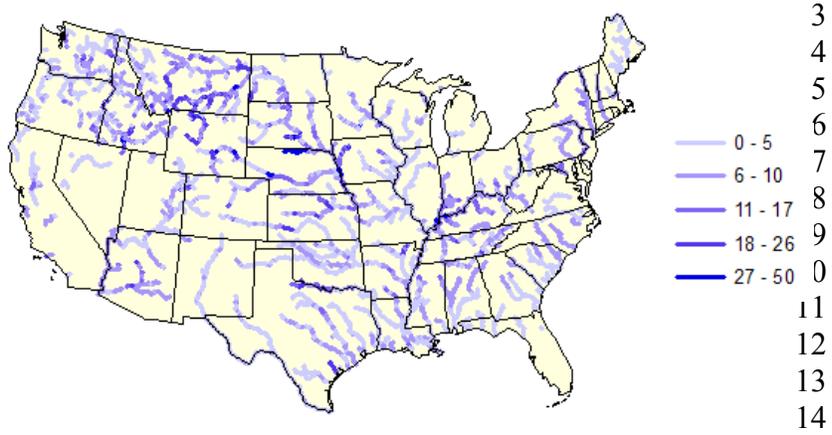
4 Photolysis (Reaction 5) dominates, but a large enough fraction of the N₂O reacts with O(¹D) that
5 this is the main source of NO_x in the stratosphere. The fate of this oxidized nitrogen (NO_y) is
6 transport back into the troposphere where it is removed by wet deposition. Downward transport
7 of the odd N from the oxidation of N₂O is a minor (~1%) source of NO_y in the troposphere.
8 Most of the N₂O released into the atmosphere is eventually converted to N₂ – the problem is that
9 it destroys stratospheric ozone in the process.

10
11 In summary, our current understanding of the chemistry of atmospheric ammonia suggests that *in*
12 *situ* conversion to N₂ and N₂O plays a minor (~1%) role in global N budgets, but if assumptions
13 about kinetics or concentrations are in error these mechanisms could become important.

14 15 16 **B. SPARROW Model for Estimating Watershed Nr** 17 18

19 Estimates of Nr transfers in aquatic ecosystems are difficult to quantify at the national scale,
20 given the need to extrapolate information from sparse monitoring data in specific watersheds to
21 the geographic boundaries of the nation. One excellent tool for estimating Nr loads at regional
22 scales is the spatially referenced regression on watershed attributes (SPARROW) modeling
23 technique. The SPARROW model has been employed to quantify nutrient delivery from point
24 and diffuse sources to streams, lakes, and watershed outlets at the national scale (Smith et al.
25 1997). The model infrastructure operates in a geographic framework, making use of spatial data
26 to describe sources of pollutants (e.g., atmospheric deposition, croplands, fertilizers) and
27 characteristics of the landscape that affect pollutant transport (e.g., climate, topography,
28 vegetation, soils, geology, and water routing). Though empirical in nature, the SPARROW
29 modeling approach uses mechanistic formulations (e.g., surface-water flow paths, first-order loss
30 functions), imposes mass balance constraints, and provides a formal parameter estimation
31 structure to statistically estimate sources and fate of nutrients in terrestrial and aquatic
32 ecosystems. The spatial referencing of stream monitoring stations, nutrient sources, and the
33 climatic and hydrogeologic properties of catchments to stream networks explicitly separates
34 landscape and surface-water features in the model. This allows nutrient supply and attenuation to
35 be tracked during water transport through streams and reservoirs, and accounts for nonlinear
36 interactions between nutrient sources and watershed properties during transport. The model
37 structure and supporting equations are described in detail elsewhere (Smith et al. 1997,
38 Alexander et al. 2000, Alexander et al. 2008). Table 3-1 provides an estimate of contemporary
39 Nr loading in surface waters of the USA, representing long-term average hydrological conditions
40 (over the past 3 decades). There are hot spots of high Nr yields to rivers (Figure 3-17) associated
41 with land use and watershed characteristics, and SPARROW allows considerations of the fate of
42 these Nr inputs to streams and rivers as they flow downstream to coastal receiving waters
43 (Alexander et al. 2008).
44

1 Figure 3-17. Total Nr yields (kg/ha/yr) in large rivers of the USA
2



15
16
17
18
19
20
21
22

3.4 Impacts and Metrics for Reactive Nitrogen

3.4.1 Measurement of Nitrogen in the Environment

Although nitrogen is among the most abundant elements on earth, only a small fraction, reactive nitrogen (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 nitrogen is transformed once introduced into the environment. Measurement methods are typically expressed in terms of mass loadings or concentrations of a particular form of nitrogen, e.g. ppm NO_x, mg/l total ammonia, or kg/ha of NO₃⁻. However, it is clear that, in addition to these metrics, there is a need to measure, compute, and report the total amount of reactive nitrogen 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 R3-24 Accordingly, the Committee recommends that the EPA routinely account for the presence of reactive nitrogen in the environment (air, land, and water) and that accounting documents be produced and published periodically (e.g. NADP summary reports).

3.4.2 Reactive Nitrogen Impacts

3.4.2.1 Classical 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 2-1 the impacts of a given source of Nr can be multiple as it 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.

The 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 reactive nitrogen 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.

Classical impacts include categories 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

1 derived that are protective of human health and the environment, the principal mission of the
 2 USEPA. This approach also encourages evaluation of damage from multiple sources as long as
 3 the characterization metric used is genuinely representative of the impact of a given contaminant.
 4 Thus, for example, the total impact of acidic gases such as SO₂ and NO_x on the acidification of
 5 watersheds can be expressed as a common metric. However, metrics for human health are
 6 generally not as simple to characterize nor are the appropriate end points; thus, the mechanism of
 7 toxicity, number of individuals affected, value of lost workdays, medical treatment costs, and
 8 value of human lives lost may all be used.

9
 10 *3.4.2.2 Ecosystem Functions and Services*

11
 12 A complementary approach to classical impact characterizations is the use of ecosystem
 13 “service” and “function” categories, in which the impairment of a specific service provided by
 14 one or more ecosystems, or function operating within an ecosystem, by causative contaminant
 15 emissions is assessed (Costanza 1997; WRI 2005). Such an approach is inherently attractive
 16 because of its basis in scientific reality, i.e. the health of humans is inextricably linked to the
 17 health of the environment. Less clear, in some cases, are ways in which to measure and monitor
 18 such impacts. Table 3-21 provides examples of ecosystem services and corresponding functions.

19
 20 **Table 3-21 Ecosystem Service and Corresponding Function Categories** (Costanza et al.
 21 1997)
 22

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

23
 24 *3.4.2.3 Economic Measures and Impacts*

25
 26 It is also possible to translate the implications of reactive nitrogen into economic terms. Two
 27 economic measures that are used are the dollar costs of damages, and the cost of remediation or

1 substitution. Another important economic metric is the cost/ton of remediation for each form of
2 reactive nitrogen. Damage costs do not always scale as tons of reactive nitrogen released into the
3 environment. If damage costs rather than tons of nitrogen were utilized as a metric, the full
4 implications of the cascade, and the setting of priorities for intervention might differ.

5
6 It is important to note that the choice of metric used in assessing impacts may play an influential
7 role in what and how one manages. Air and water protection laws state that the goal is “to protect
8 human health and the environment.” Yet, there is no generally agreed upon common metric for
9 measuring the full range of effects (which are complex and often unknown), or for setting
10 priorities in the establishment or implementation of policies.

11
12 As noted above, there are multiple metrics for measuring reactive nitrogen or any other agent in
13 the environment. The most common metric utilizes quantitative measures of the total amount of
14 reactive nitrogen (and any of its specific chemical forms) in different environmental reservoirs
15 and the mass flux between them. But while providing common units, typically mass or
16 concentration, these gross measures do not distinguish the relative societal costs of health or
17 environmental consequences of reactive nitrogen of different forms or places in the cascade. Two
18 economic measures that are used are the dollar costs of environmental or health damages, and the
19 cost of mitigation or remediation. While not all damages can be turned into economic costs (see
20 Chesapeake Bay box), and the costs of some damages have not been quantified, enough of the
21 major damages can be quantified economically for this to provide a useful complementary metric
22 for decision-making.

23
24 The advantage of monetizing damages is that it reflects an integrated value that human society
25 places on lost ecosystem goods and services in common currency, and illustrates the cascading
26 costs of damages as reactive nitrogen changes form and moves between different parts of the
27 ecosystem. In addition, human health implications can also be included as the cost of health care
28 treatment, lost work days and other aspects of morbidity and mortality (e.g., economic value of
29 lives lost). A third metric is to look at morbidity and/or mortality separately, and not monetize
30 them with a cost value. Of course a concern, particularly with respect to the economic metric, is
31 that there are a number of ecosystem services that arguably cannot be easily monetized, for
32 example the loss of biodiversity, and those ecosystem functions that affect climate change –
33 particularly those considered to be regulating and supporting ecosystem services (as defined by
34 the Millennium Ecosystem Assessment) are particularly difficult to fit into an economic metric.
35 It is thus essential that a variety of complementary metrics be used to assess the impact of
36 anthropogenic reactive nitrogen on the environment and human well being.

37 38 *3.4.2.4 Summary of Nr Metrics*

39
40 There is value in each of the ways that nitrogen metrics are expressed. Traditional categories
41 provide a readily adaptable framework for regulation, while ecosystem service and function-
42 based categories provide a richer context for stating the complex connections among reactive
43 nitrogen inputs and transformations and their impacts on ecosystem health and human well-
44 being. Dollar-based metrics provide a means of identifying those effects that have the greatest
45 effects on the costs to society.

1 **Finding F3-16.** The Committee finds that reliance on only one approach for categorizing the
 2 measurement of reactive nitrogen is unwise, and unlikely to result in the desired outcome of
 3 translating nitrogen-induced degradation into the level of understanding needed to develop
 4 support for implementing effective nitrogen management strategies.

5
 6 **Recommendation F3-25.** It is recommended that the EPA examine the full range of impact
 7 categories as a basis for expressing Nr impacts in the environment, and for building better
 8 understanding and support for integrated management efforts.
 9

10

Chesapeake Example

Example: Economic Impacts and Metrics for Chesapeake Bay

Recently, the nitrogen 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 reactive nitrogen fluxes in the Chesapeake Bay water and air shed in Figure CB-1 below. Note that approximately 48% of nitrogen entering the watershed is coming through emissions to the atmosphere, but they are causing 65% of the dollar damages and 88% of the mortality. A nearly equal percentage, 49%, of the reactive nitrogen involves runoff from the land, but it accounts for only 26% of the damage costs and 12% of the mortality. Fresh water releases of reactive nitrogen account for only 3% of the reactive nitrogen 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 nitrogen mitigation provide an additional economic measure of the cost effectiveness of actions to reduce a ton of nitrogen.

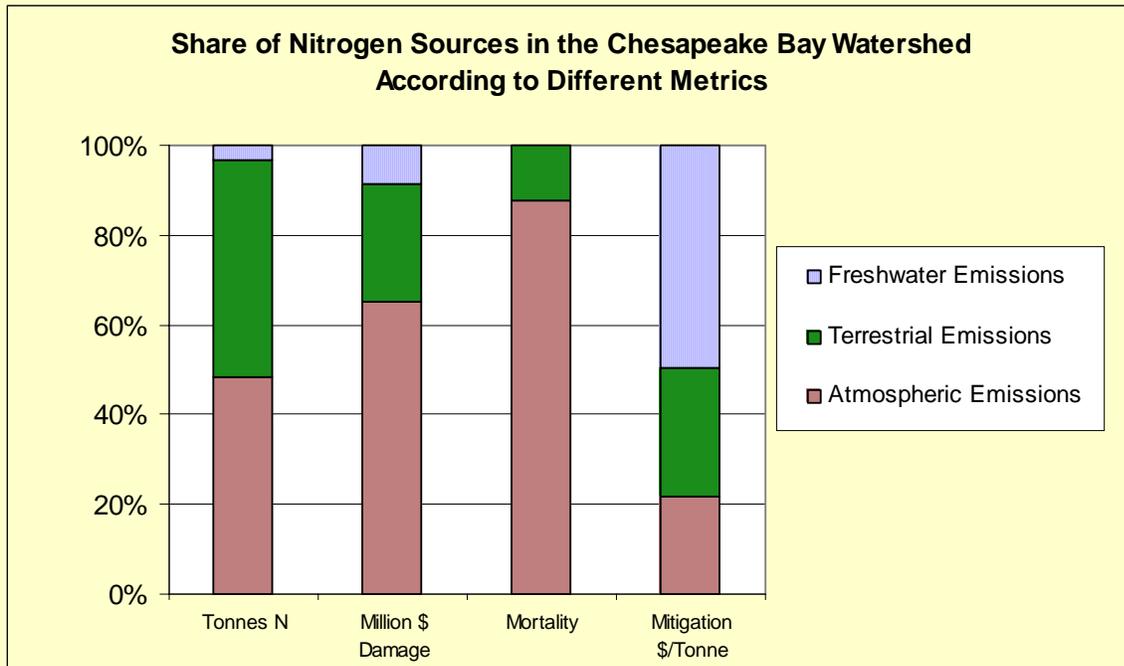


Figure CB-1. Relative importance of all reactive nitrogen sources in the Chesapeake Bay Watershed according to four different metrics.

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 3.2.

Table CB-1. Alternative metrics for different atmospheric emissions and for terrestrial and freshwater releases of reactive NO_x and NO by source.

	Reactive nitrogen (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 reactive nitrogen releases from the land even though the two sources are comparable in scale in terms of reactive forms of nitrogen 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 CB-2.

Looking at the cost of remediation of reducing releases to the environment, the least costly per ton of reactive nitrogen or per dollar saved is also to be found in the case of atmospheric emissions. While most legislation does not allow the issue of cost for remediation to be considered, it may be 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.

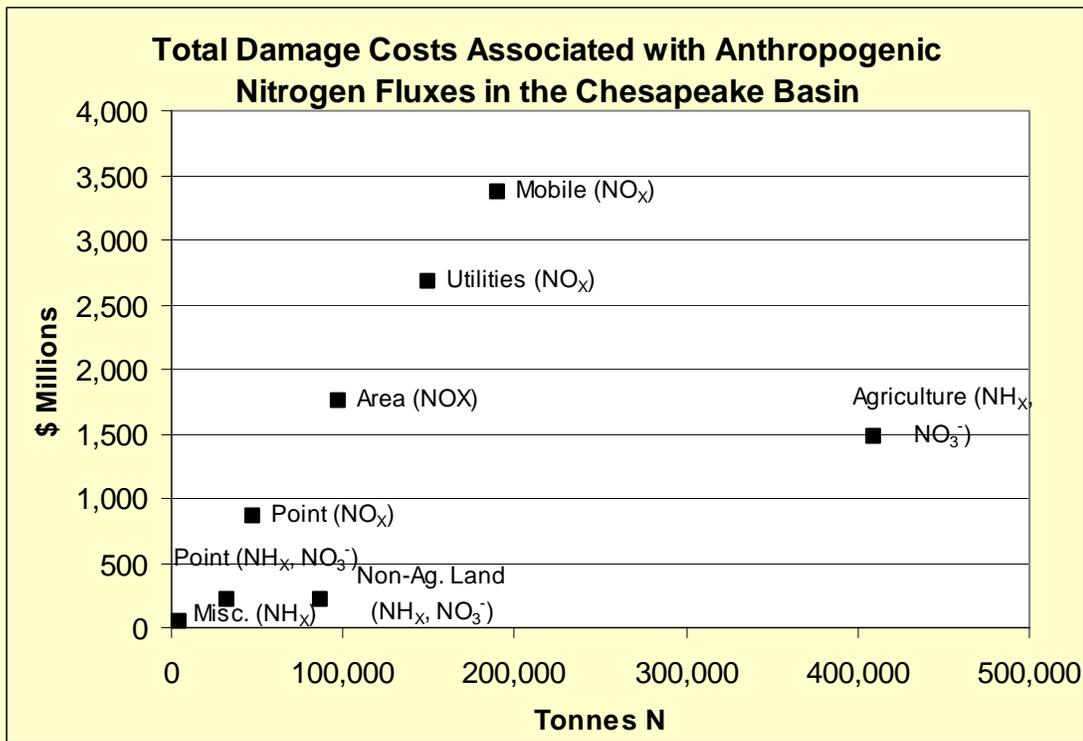


Figure CB-2. Scatter plot of all quantifiable damage costs (including health impacts) relative to tons of reactive nitrogen showing the significant difference in emphasis of the two different metrics.

Similarly, economic losses due to climate change and ozone depletion from N₂O 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 reactive nitrogen 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 health as well as their large contribution to the degradation of Chesapeake Bay water quality. Hence, if one is interested in reducing water impacts of reactive nitrogen, the total reduction of damage may rely nearly as much on stricter enforcement of the clean air act as the clean water act. This challenges our traditional approach to regulation, but that is a consequence of examining reactive nitrogen through the nitrogen cascade.

1
 2 **3.4.3 Nitrogen Impacts on Aquatic Ecosystems**

3
 4 The availability of nitrogen controls primary production in much of the world's estuarine, near-
 5 shore coastal and open ocean waters (Dugdale 1967, Ryther and Dunstan 1971, Nixon 1995,
 6 Paerl 1997; Boesch et al. 2001). Nitrogen can also play a role as either a primary or secondary

1 limiting nutrient in freshwater environments, especially large lakes (e.g. L. Tahoe, L. Superior).
2 As such, the fertility of these waters is often closely controlled by N inputs, which are provided
3 either internally by regeneration of pre-existing N and biologically-fixed atmospheric N₂, or
4 supplied externally (i.e. “new” N) as combined N sources delivered via surface runoff, sub-
5 surface groundwater or atmospheric deposition.
6

7 During the past century, following large-scale use of synthetic N fertilizers in agriculture, rapid
8 expansion of industrial and transportation-related fossil fuel combustion and coastal
9 urbanization, humans have significantly altered the balance between “new” N inputs and N
10 losses in the marine environment (Codispoti et al. 2001). During this time frame, terrigenous
11 discharge and atmospheric N emissions have increased by 10 fold (Howarth et al. 1996, Holland
12 et al., 1999). For at least 50 years, researchers have recognized this growing imbalance,
13 especially in estuarine and coastal waters where anthropogenically driven N_r over-enrichment
14 has fueled accelerated primary production, or eutrophication (Vollenweider et al. 1992, Nixon
15 1995).
16

17 Eutrophication is a condition where nutrient-enhanced primary production exceeds the ability of
18 higher ranked consumers and organic matter-degrading microbes to consume and process it.
19 D’Elia (1987) characterized this condition as “too much of good thing”, or over-fertilization of
20 N-limited marine ecosystems with “new” N, a bulk of it being anthropogenic (Howarth et al.
21 1986, Vitousek et al. 1997, Galloway and Cowling 2002). Symptoms of N-driven eutrophication
22 vary from subtle increases in plant production to changes in primary producer community
23 composition, to rapidly accelerating algal growth, visible discoloration or blooms, losses in water
24 clarity, increased consumption of oxygen, dissolved oxygen depletion (hypoxia), which is
25 stressful to resident fauna and flora, or in the case of total dissolved oxygen depletion (anoxia),
26 elimination of habitats (Paerl 1988, 1997, Diaz and Rosenberg 1995, Rabalais and Turner 2001;
27 Diaz and Rosenberg 2008). Other effects include submerged aquatic vegetation (SAV) losses,
28 possible impacts on tidal wetland health, and disruption of estuarine food chain dynamics that
29 may favor an imbalance towards lower trophic levels (e.g., jellyfish) or microbial looping.
30

31 The extent to which accelerated N loading promotes eutrophication and its symptoms varies
32 greatly among marine ecosystems. Receiving waters exhibit variable sensitivities to N and other
33 nutrient (P, Fe, Si) loads that are controlled by their size, hydrologic properties (e.g. flushing
34 rates and residence times), morphologies (depth, volume), vertical mixing characteristics,
35 geographic and climatic regimes and conditions. In addition, the magnitude and distribution of
36 N in relation to other nutrient loads can vary substantially. In waters receiving very high N loads
37 relative to requirements for sustaining primary and secondary production, other nutrient
38 limitations may develop. This appears to be the case in coastal waters downstream of rivers
39 draining agricultural regions that are enriched in N.
40

41 On the ecosystem level, estuarine and coastal waters exhibit individualistic responses to N loads
42 over seasonal and longer (multi-annual, decadal) time scales. The degree to which these systems
43 are exposed to freshwater discharge, tidal exchange and vertical mixing is critical for
44 determining how they respond to specific N loads (Vollenweider et al., 1992, Nixon 1995,
45 Cloern 1999, 2001; Valdes-Weaver et al. 2006; Paerl et al. 2007). Another variable is the
46 manner in which N loading takes place, which may range from acute pulsed events such as

1 storms and associated flooding, to longer-term gradual (chronic) increases in N loading
2 associated with more predictive seasonal, annual and inter-annual hydrologic cycles. There are
3 striking contrasts in ecosystem response to N inputs that reflect a range in physical
4 (hydrodynamic, optical) and climatic conditions (Cloern 1999, 2001). Examples include
5 contrasts between strong tidally-driven estuarine systems such as Delaware Bay and San
6 Francisco Bay, and non-tidal, lagoonal systems such as North Carolina's Pamlico Sound and
7 Texas's Laguna Madre, or semi enclosed coastal systems such as Florida Bay and the Long
8 Island Sound (Bricker et al., 1999; Valdes-Weaver et al. 2006; Paerl et al. 2007).

9
10 Externally-supplied N comes in various forms, including organic N and inorganic reduced
11 (ammonia and ammonium ion) and oxidized (nitrate, nitrite) N, all of which are potentially
12 available to support new production and eutrophication. Laboratory experiments on
13 phytoplankton isolates and bioassays with natural phytoplankton communities have indicated
14 that these contrasting forms may be differentially and preferentially utilized, indicating that,
15 depending on composition of the affected phytoplankton community, some forms are more
16 reactive than others (Collos, 1989; Stolte et al., 1994; Riegman, 1998). Phytoplankton
17 community composition can also be altered by varying proportions and supply rates of different
18 forms of N (Dortch, 1990; Stolte et al., 1994; Harrington, 1999; Pinckney et al., 1999; Piehler et
19 al., 2002). In addition, specific N compounds may interact with light availability,
20 hydrodynamics and other nutrients, most notably P, Si, Fe, and trace metals, to influence
21 phytoplankton community growth rates and composition (Harrison & Turpin, 1982; Smith, 1990,
22 Dortch & Whitley, 1992).

23
24 One example of shifting N inputs is the proliferation of intensive livestock operations in coastal
25 watersheds, which has led to large increases and changes in chemical composition of nitrogenous
26 compounds discharged to estuarine and coastal waters via runoff, groundwater and atmospheric
27 deposition (Paerl, 1997; Howarth, 1998; Galloway & Cowling, 2002). In general, coastal waters
28 under the influence of these operations are experiencing increases in total N loading as well as a
29 shift toward more reduced N (ammonium, organic N) relative to oxidized N (nitrate, nitrite)
30 (Howarth et al., 2002; Galloway & Cowling, 2002). These increases, combined with increases in
31 hypoxia and anoxia in receiving waters, are leading to more ammonium-rich conditions, which
32 will favor algal groups able to best exploit this N form, including some harmful algal bloom
33 (HAB) taxa (Paerl and Whitall 1999; Paerl et al. 2007). Similarly, conversion of forest and
34 agricultural lands to urban lands can alter landscapes and promote nitrogen loading to estuaries
35 by increasing impervious pathways and removing natural landscape filters for N. Development
36 also destroys and eliminates wetlands, leading to more nitrate-enriched conditions, potentially
37 favoring plant taxa best able to exploit this N form.

38 39 *3.4.3.1 Water Quality Regulation and Management*

40
41 Traditionally, reactive nitrogen and other land, air and water pollutants are measured in terms of
42 quantity (mass) released per operation *in toto*, or in concentration terms. Regulations often
43 specify mass limits on releases in tons, or else maximum concentrations allowed in air or water.
44 Under Section 303 of the Clean Water Act (CWA), states are required by the EPA to adopt water
45 quality standards and criteria that meet the state-identified designated uses (e.g., uses related to
46 "fishable", "swimmable") for each waterbody. Specifically, "A water quality standard defines

1 the water quality goals of a water body, or portion thereof, by designating the use or uses to be
2 made of the water and by setting criteria necessary to protect the uses. States adopt water quality
3 standards to protect public health or welfare, enhance the quality of water and serve the purposes
4 of the Clean Water Act (the Act).” (40 CFR Sec. 131.2). Further, “Such standards serve the dual
5 purposes of establishing the water quality goals for a specific water body and serve as the
6 regulatory basis for the establishment of water-quality-based treatment controls and strategies
7 beyond the technology-based levels of treatment required by sections 301(b) and 306 of the
8 Act.” (40 CFR Sec. 131.2).

9
10 The EPA sets minimum requirements for approvable standards and criteria: Among the elements
11 that must be included in each State’s water quality standards submitted to EPA for review are:
12 use designations; water quality criteria sufficient to protect the designated uses; and an
13 antidegradation policy (40 CFR Sec. 131.6). If minimum requirements are not met by the states
14 within the allowable timeframe for state standards adoption, the EPA has the authority to set
15 state standards where they may be lacking and the EPA-promulgated standards then become
16 applicable. States are further required to update their standards and criteria on a triennial basis,
17 including a public hearing to provide for public input.

18
19 In the mid to late 1990s, given the extent and prevalence of nutrient related impairments, EPA
20 began to emphasize the development of numeric nutrient criteria for both phosphorus and
21 nitrogen through the state standards-setting process. According to the 1996 Water Quality Report
22 to Congress (EPA 1997), 40% of the rivers, 51% of the lakes and ponds, and 57% of the
23 estuaries assessed for the report were exhibiting a nutrient-related impairment. In particular, the
24 impact of estuarine hypoxic zones in Chesapeake Bay, the Gulf of Mexico and Long Island
25 Sound, among others, emphasized the need for added attention. Since few states had adopted
26 numeric nutrient criteria for all affected waterbodies, especially for nitrogen, often relying on
27 narrative criteria or secondary effects such as chlorophyll-a concentration, dissolved oxygen, or
28 water clarity, EPA’s strategy, driven by President Clinton’s Clean Water Action Plan (EPA,
29 1998) mandated numeric nutrient criteria to begin to address the problem (e.g. Total Maximum
30 Daily Loads or TMDLs for nitrogen) (EPA 1999).

31
32 To move the objectives of the Clean Water Action Plan forward, EPA published national nutrient
33 criteria guidance for lakes and reservoirs (EPA 2000b), rivers and streams (EPA 2000c),
34 estuaries and coastal waters (EPA 2001c) and wetlands (EPA 2007b), and ecoregional guidance
35 for lakes and reservoirs and rivers and streams (Figure 3-19 and Table 3-22). Regional
36 assessment groups were formed to develop criteria for nitrogen, phosphorus and related
37 parameters and approaches for potential adoption into state water quality standards consistent
38 with the EPA guidance. While states have considerable latitude in how they choose to formulate
39 and adopt nutrient criteria into their standards, the CWA and federal regulations require that they
40 be at least as protective as the criteria EPA proposes in their guidance and criteria documents.

41
42 To date, relatively few states have adopted new numeric criteria into their water quality
43 standards. While some successes are evident in promulgating phosphorus criteria for freshwater
44 systems, which has a richer history of numeric criteria incorporation into state water quality
45 standards, development of numeric nitrogen criteria has been elusive for a variety of reasons.
46 First, the role of nitrogen as a limiting nutrient in freshwaters is not well known, although recent

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1 evaluations suggest a more prominent role of nitrogen in freshwater enrichment (Dodds, 2006;
2 Dodds, Smith and Lohman, 2002). Setting nitrogen criteria for freshwaters without a thorough
3 understanding of effects and benefits of management could lead to overly stringent or
4 underprotective criteria. Second, the EPA criteria guidance have emphasized an ambient nitrogen
5 concentration as a criterion, using data from clean, reference sites on a regional basis to set
6 criteria. While states can choose to adopt site-specific and mass loading criteria, individual water
7 body conditions and other complexities render it difficult to set reasonable and protective
8 nitrogen loads without additional study or modeling efforts, which could take years. Setting
9 criteria is further complicated by the defined endpoint, or designated use, which might more
10 appropriately be considered impaired due to excessive chlorophyll concentrations, low oxygen
11 levels (hypoxia) or biological impacts (HABs), rather than an ambient concentration of nitrogen.
12 Third, there are concerns about when and how to measure nitrogen to reflect system health and
13 whether the criteria will be attainable under today's landscape. This is further compounded by
14 multimedia and interstate, or even international, sources of nitrogen as it blows or flows across
15 jurisdictional boundaries in air and water.

16
17 The CWA has little authority over atmospheric sources, and individual states explicitly lack
18 authority to control upstream sources that are outside of their jurisdictional boundaries. Quite
19 often in estuaries such as the Gulf of Mexico or Chesapeake Bay, management goals that meet
20 water quality standards cannot be attained without interstate compacts or a strong federal role
21 that may be resisted by upstream states that may have to bear the cost but do not necessarily reap
22 the benefits of the water quality improvement. Such a dilemma underscores the need for an
23 integrated approach to Nr management.

24

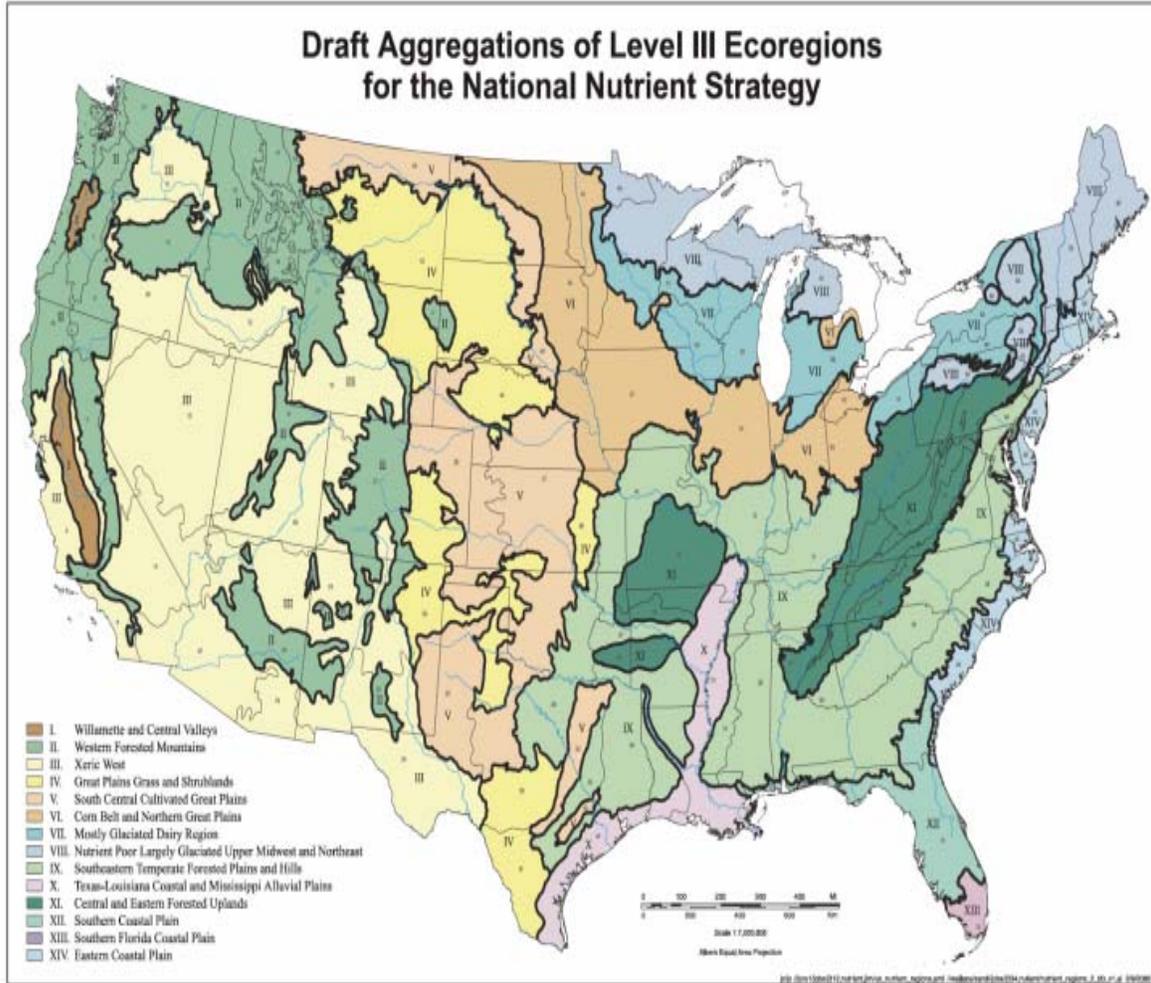


Figure 4. Fourteen nutrient ecoregions as delineated by Omernik (2000). Ecoregions were based on geology, land use, ecosystem type, and nutrient conditions.

1
2 Note: this figure needs to be renumbered as Figure 3-18
3
4
5

1
2

Table 3-22 Nutrient Criteria for Ecoregions (EPA 2000)

Parameter	Aggregate Ecoregion (See Figure 3-19)													
	I	II	III	IV	V	VI	VII	VIII	IX	X	XI	XII	XIII	XIV
Lakes and Reservoirs														
TP (ug/L)		8.75	17.00	20.00	33.00	37.5	14.75	8.00	20.00		8.00	10.00	17.50	8.00
TN (mg/L)		0.10	0.40	0.44	0.56	0.78	0.66	0.24	0.36		0.46	0.52	1.27	0.32
Chl <i>a</i> (ug/L)		1.90	3.40	2.00 ¹	2.30 ¹	8.59 ¹	2.63	2.43	4.93		2.79 ¹	2.60	12.35 ²	2.90
Secchi (m)		4.50	2.70	2.00	1.30	1.36	3.33	4.93	1.53		2.86	2.10	0.79	4.50
Rivers and Streams														
TP (ug/L)	47.00	10.00	21.88	23.00	67.00	76.25	33.00	10.00	36.56	128 ⁴	10.00	40.00		31.25
TN (mg/L)	0.31	0.12	0.38	0.56	0.88	2.18	0.54	0.38	0.69	0.76	0.31	0.90		0.71
Chl <i>a</i> (ug/L)	1.80	1.08	1.78	2.40	3.00	2.70	1.50	0.63	0.93 ¹	2.10 ¹	1.61 ¹	0.40 ¹		3.75 ¹
Turb (FTU)	4.25	1.30 ³	2.34	4.21	7.83	6.36	1.70 ³	1.30	5.70	17.50	2.30 ³	1.90 ³		3.04
¹ Spectrophotometric method. ² Trichromatic method. ³ NTU ⁴ Inordinately high value – may be a statistical anomaly or reflective of a unique condition. Further investigation warranted. Source: U.S. EPA, http://www.epa.gov/waterscience/criteria/nutrient/ecoregions/files/sumtable.pdf														

3
4
5

3.4.3.2 Water Management in Urbanized Areas

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

21
22
23

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

- 24 • Prevention or source controls
- 25 • Physical, chemical or biological “dead ending” or storage within landscape compartments
 26 where it is rendered less harmful (e.g., long-term storage in soils or vegetation;
 27 denitrification, primarily in wetlands; reuse)
- 28 • Treatment using engineered systems such as STPs or best management practices (BMP) for
 29 stormwater and nonpoint source runoff.

1 While most management programs focus on the third (treatment) approach, there are opportunities
2 for combining the three that can be more effective and cost less.

3
4 To better address Nr runoff and discharges from the peopled landscape the INC recommends that
5 EPA:

6
7 **Recommendation R3-26.** Evaluate the suite of regulatory and non regulatory tools used to
8 manage Nr in populated areas from nonpoint sources, stormwater and domestic sewage and
9 industrial wastewater treatment facilities, including goal-setting through water quality standards and
10 criteria. Determine the most effective regulatory and voluntary mechanisms to apply to each source
11 type with special attention to the need to regulate nonpoint source and related land use practices.

12
13 **Recommendation R3-27.** Review current regulatory practices for point sources, including both
14 wastewater treatment plants and stormwater, to determine adequacy and relationship towards
15 meeting national Nr management goals. Consider technology limitations, multiple pollutant
16 benefits, and funding mechanisms as well as potential impacts on climate change from energy use
17 and greenhouse gas emissions, including nitrous oxide.

18
19 **Recommendation R3-28.** Set Nr management goals on a regional/local basis, as appropriate, to
20 ensure most effective use of limited management dollars. Fully consider “green” management
21 practices such as low impact development and conservation measures that preserve or re-establish
22 Nr removing features to the landscape as part of an integrated management strategy along with
23 traditional engineered best management practices.

24
25 **Recommendation R3-29.** Research best management practices that are effective in controlling
26 Nr, especially for nonpoint and stormwater sources, including land and landscape feature
27 preservation and set Nr management targets that realistically reflect these management and
28 preservation capacities. Construct a decision framework to assess and determine implementation
29 actions consistent with management goals.

30
31 **Recommendation R3-30.** Use ecosystem-based management approaches that balance natural and
32 anthropogenic needs and presence in the landscape.

33 34 *3.4.3.3 Water Quality Monitoring and Assessment*

35
36 Fundamental to a successful state water quality management program are regular monitoring and
37 periodic assessment of conditions that identify waters that are impaired. Under Sec. 106 of the
38 CWA, the EPA provides funds to assist state and interstate agencies and tribes to conduct
39 monitoring of the nation’s waters to ensure adopted water quality criteria, and designated uses,
40 are met. Further, primarily under Sec. 305(b) of the CWA, those entities are required to report,
41 on a biennial basis, on the health and status of their jurisdictional waters.

42
43 These assessments are presented by the states as categories of support of designated uses to the
44 EPA. Past practice was for EPA to compile the reports and information provided by the states
45 into a National Water Quality Report to Congress on the health of the nation’s waters, generally
46 referred to as a “305(b) report.” The last such report was published in 1998 (EPA 2000a), after

1 which it transitioned into a Water Quality Report in 2000 (EPA 2002) and a National
2 Assessment Database in 2002 (<http://www.epa.gov/waters/305b/index.html>) as EPA sought more
3 expeditious ways to present useful information to Congress and the public. Subsequent reports
4 will provide a synthesis of CWA Sec. 305(b) and 303(d) reporting under a Consolidated
5 Assessment and Listing Methodology or “CALM” approach.

6
7 Requirements of Sec. 303(d) of the CWA provide another key mechanism for states and tribes to
8 report on water quality, focusing on identified “impaired” waters. Impaired waters are those that
9 do not meet identified designated uses, generally because they do not meet either numeric or
10 narrative criteria or otherwise do not support designated use categories for other reasons. These
11 waters are compiled into a list, i.e., the “303(d) list”, which identifies those waters for which a
12 Total Maximum Daily Load or “TMDL” needs to be developed (EPA 1999). A TMDL
13 identifies the pollutant causing the impairment and how much it must be reduced to meet state
14 water quality standards and allocates the reductions between point sources (the “wasteload
15 allocation”) and nonpoint sources (the “load allocation”). The EPA has rights of approval over
16 state 303(d) lists, and all listed waters must be categorized for immediate action or for addressing
17 at a future time using an assignment of priorities based on many factors including the severity of
18 the problem and the threat it poses to human health and the environment.

19
20 The EPA compiles the approved state 303(d) lists into a national listing
21 (http://iaspub.epa.gov/waters/national_rept.control). The list provides information by state as
22 well as by impairment cause, and identifies the TMDLs completed to date. The most current data
23 available on the EPA website includes reporting from most entities through 2004. The report
24 identifies 5617 impairments related to “nutrients” (almost 9% of all identified impairments),
25 although other impairments may ultimately have a nutrient enrichment cause. For example,
26 oxygen depletion (4540), turbidity (2050), algal growth (510), ammonia (generally toxicity –
27 416), and harmful algal blooms (HABS) (4) can all have a common cause such as nitrogen or
28 phosphorus enrichment. It should also be clear that impairments may have multiple causes so, for
29 example, waters identified as impaired by oxygen depletion may also be impaired by nutrients.

30
31 There are other initiatives promoted by EPA to monitor and assess the nation’s waters, generally
32 implemented in collaboration with, or by, the state and interstate agencies and tribes having
33 jurisdiction over the waters. These include the Wadeable Stream Assessment (WSA) (EPA
34 2006a), the National Coastal Assessment (NCA) and its National Coastal Condition Reports
35 (EPA 2001a, 2004, and 2006b), the Survey of the Nation’s Lakes and Survey of the Nation’s
36 Rivers and Streams, and, more recently, probabilistic monitoring efforts in lakes, streams and
37 estuaries (<http://www.epa.gov/owow/monitoring/reporting.html>). Many of these are aimed at
38 including a biological assessment component that is often lacking in water pollutant and
39 chemistry efforts described above.

40
41 The WSA is designed to evaluate aquatic community structure of macroinvertebrates to assess
42 “aquatic life use” support of flowing waters that are shallow enough to wade in. The WSA is
43 implemented by state environmental agencies, and the program has matured over the years since
44 its inception to relate probable causes of impaired biological communities and inclusion of those
45 impairments on state 303(d) lists. A national summary of state assessments of wadeable streams
46 (EPA 2006a) identified only 28% of the streams assessed to be in good condition while 25%

1 were rated fair and 42% poor. These effects are related to a variety of stressors, which may
2 include nutrient effects, but the assessments are not generally specific as to cause, instead
3 reflecting the cumulative effect of “pollution” including physical and chemical insult.
4 The NCA National Coastal Condition Reports (EPA 2001a, 2004 and 2006b) are more closely
5 related to nutrient enrichment assessments, especially for manifestations of nutrient enrichment
6 such as hypoxia, nuisance algal blooms, and general habitat degradation. The last
7 comprehensive national NCCR was published in 2004 (EPA 2004) with a more recent
8 assessment focused on 28 National Estuary Program estuaries published in 2007 (EPA 2006b).
9 The 2004 NCCR included an overall rating of “fair” for estuaries, including the Great Lakes,
10 based on evaluation of over 2000 sites. The water quality index, which incorporates nutrient
11 effects primarily as chlorophyll-*a* and dissolved oxygen impacts, was also rated “fair” nationally.
12 Forty percent of the sites were rated “good” for overall water quality, while 11% were “poor”
13 and 49% “fair”. An indicator based on dissolved inorganic nitrogen (DIN) samples collected
14 during the sampling surveys reported in 2004, contrary to overall water quality, categorized only
15 5% of the sites, primarily in the Northeastern United States, as “poor”, and 13% as “fair”. This
16 may highlight the difficulty with ambient nutrient concentration criteria, which may not
17 adequately reflect the symptoms of enrichment unless monitoring is intensive and properly
18 timed.

19
20 The National Oceanic and Atmospheric Administration (NOAA) has periodically produced
21 estuarine assessments under the National Estuarine Eutrophication Assessment (NEEA)
22 program. The most recent report was released in 2007 (Bricker et al., 2007). The report has a
23 focus on nutrient enrichment and its manifestations in the estuarine environment and relies on
24 participation and interviews of local experts to provide data for the assessment. Among the key
25 findings were:

- 26
27 • Eutrophication is a widespread problem, with the majority of assessed estuaries
28 showing signs of eutrophication—65% of the assessed systems, representing 78%
29 of assessed estuarine area, had moderate to high overall eutrophic conditions.
- 30
31 • The most common symptoms of eutrophication were high spatial coverage and
32 frequency of elevated chlorophyll *a* (phytoplankton)—50% of the assessed
33 estuaries, representing 72% of assessed area, had excessive chlorophyll *a* ratings.

34
35
36 **Finding F3-17.** The Committee has determined that an integrated approach to
37 monitoring that includes multimedia (air, land and water) components and considers a
38 suite of environmental and human concerns (e.g., Nr effects, climate change, human
39 health) would be most useful and efficient. Some of the phenomena that we present in
40 this report simply need more definition and verification but, more importantly, as control
41 is brought to bear on reactive nitrogen, improvements need to be measured (i.e.
42 monitored) to validate the success of one control or another. If the desired improvements
43 are not realized as shown by the collected data, corrective measures will be required. The
44 pool of data would used to formulate new management procedures. The process of
45 monitoring and control revisions is termed adaptive management—a process that the
46 committee supports as it does not delay actions that can be taken now, but acknowledges

1 the likelihood that management programs will be altered (adapted) as scientific and
 2 management understanding improve.

3
 4 **Recommendation R3-31.** Therefore, the INC recommends that EPA initiate discussions
 5 and take action to develop a national, multimedia monitoring program that monitors
 6 sources, transport and transition, effects using indicators where possible, and sinks of Nr.
 7 This comprehensive program should build upon existing EPA and state initiatives as well
 8 as monitoring networks already underway in other federal agencies such as the USGS
 9 programs and the NADP effort.

10
 11 *3.4.3.4 Attainment of Water Quality Management Goals and Standards*

12
 13 Sections 3.4.3.1 and 3.4.3.2 have identified the water quality impacts caused by excess Nr in the
 14 environment. As indicated, estuarine systems are most often susceptible to Nr enrichment where
 15 nitrogen is more likely to be the limiting nutrient (Paerl 1997; Boesch et al. 2001). EPA through
 16 the CWA has established guidance for nutrient criteria with the expectation that states will
 17 ultimately adopt numeric criteria in state water quality standards, although the process is well
 18 behind schedule. Defining single number criteria for nutrients or related indicators representative
 19 of undesirable levels of productivity (e.g., chlorophyll *a*) is difficult, even using an ecoregional
 20 approach. To address site specific, or estuary specific, characteristics state managers more often
 21 use the formal TMDL process or collaborative estuarine management plans that set nitrogen
 22 management targets to meet existing, related water quality criteria (e.g., dissolved oxygen or
 23 chlorophyll *a*). Some of the more prominent efforts and targets for nitrogen control, are
 24 summarized in Table 3-23.

25
 26 **Table 3-23.** 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	7%/year, 2000-2010	?

27
 28
 29 These targets all exceed the potential decreases in nitrogen loadings the Committee has identified
 30 in this report from a national perspective. The Committee estimates decreases in specific source
 31 categories that are generally less than 25%. Since not all sources offer management
 32 opportunities, the expectation is that Nr loadings to estuaries would cumulatively be less than
 33 25%, which is below the targets identified in Table 3-23. Many of the management actions the
 34 Committee has proposed would also require substantive changes in national programs, regulatory
 35 authority, management technologies and societal demands to be accomplished. This is a nutrient
 36 management concern state managers are well aware of as they develop TMDLs and management
 37 plans that range above attainment potential, not only for Nr but more frequently for other

1 pollutants that are predominately nonpoint source (NPS) and stormwater (SW) loaded (including
2 atmospheric source contributions).
3

4 The Chesapeake Bay Program, for example, is a model for Nr and P management in many ways.
5 Considerable resources were committed, and many best management practices (BMPs)
6 implemented, with disproportionate results. Despite regional efforts and commitments from all
7 watershed states, and more funding than any other estuary program is likely to see, they are
8 falling short of management targets and are coming off a discouraging year (2007) that saw a
9 severe hypoxic episode. Similarly, the adoption of the Long Island Sound TMDL (See Long
10 Island Sound Text Box) sets an implementation plan that could attain CT and NY dissolved
11 oxygen criteria, but only if “alternative technologies” that have nothing to do with Nr source
12 control, such as mechanical aeration of the Sound or biological harvesting of nutrients, are used.
13

14 To meet DO criteria in LIS of “never less than” 3.0 mg/L in NY and 3.5 mg/L in CT, with
15 incremental exposure periods up to 4.8 mg/L would, according to the LIS model, require
16 nitrogen management to very near a pre-Colonial condition. While management of LIS
17 “benefits” from the dominance of point source loading (about 2/3 of the load from CT and NY
18 sources) which is relatively easy to manage, even pushing those sources to limits of technology
19 and expanding management throughout the basin to include MA, NH and VT would not attain
20 DO criteria. NPS and SW sources, including sources related to atmospheric deposition, which
21 are more difficult to control beyond a ten or twenty percent removal efficiency, and at an
22 considerable cost, preclude full attainment of DO criteria. EPA’s Clean Water Needs Survey
23 (EPA 2008) has identified more than \$200 billion in wastewater management infrastructure
24 needs that does not fully address nutrient control from both traditional point as well as
25 nonpoint/stormwater sources.
26

27 Meeting Nr management goals for estuaries, when a balance must be struck between economies,
28 society and the environment, under current federal law seems unlikely. Enforceable authorities
29 over nonpoint source, stormwater, air (in terms of critical loads), and land use – both
30 development and agriculture are inadequate to require necessary Nr controls. Funding programs
31 are presently inadequate to meet existing pollution control needs. Further, new technologies and
32 management approaches are required to meet ambitious Nr control needs aimed at restoring
33 national water quality.
34

35 **Recommendation R3-32.** Therefore, the INC recommends that EPA reevaluate water quality
36 management approaches to ensure Nr management goals are attainable, enforceable, and
37 affordable and that monitoring and research are adequate to problem definition and resolution,
38 particularly in the development of nitrogen removal technologies. This may require changes in
39 the way EPA sets criteria and some compromises in ecosystem goals to accommodate human
40 uses of the air, land and water.
41
42
43
44
45
46

1
2

Long Island TMDL Example

Example: 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 oxygen that violate both Connecticut's and New York's water quality standards. Nitrogen has been identified as the pollutant that causes substandard levels of dissolved oxygen in Long Island Sound and, accordingly, Connecticut's and New York's environmental agencies have developed a TMDL that assigns nitrogen reductions from both point sources (the wasteload allocation or WLA) and nonpoint sources (the load allocation or LA) in their respective states to meet the established 58.5% reduction of anthropogenic sources.

The Long Island Sound TMDL is set at 23,966 tons of nitrogen/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 nitrogen load comes from point sources – POTWs (publicly owned treatment works) and CSOs (combined sewer overflows) – accounting for 38,899 tons/yr of the total nitrogen 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 oxygen criteria in both states. The revised TMDL will likely require more aggressive reductions of nitrogen to meet dissolved oxygen criteria and may formalize targets for upstream state contributions and atmospheric deposition.

3

3.4.4 CAA 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, nitrogen dioxide, ozone, sulfur dioxide, and particulate matter. 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 NO₂ is the only reactive nitrogen 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.

Control of air pollution involves federal-State partnerships. Any State in violation of the NAAQSs is required to develop a State Implementation Plan (SIP) for each pollutant exceeding the standard. These SIPs require monitoring and analysis of ambient concentrations and trends as well numerical modeling of current and future pollution levels. If the EPA approves the SIP, it delegates to that State federal authority to implement and enforce regulations. The implementation strategies may involve emissions controls on stationary sources, regulation of consumer products such as oil-based paints, vehicular traffic control, public transportation systems, as well as testing, inspection and maintenance of motor vehicles. Agricultural practices are generally not included as part of the plan.

The CAA was amended “to provide for a more effective program to improve the quality of the nation’s air” in 1970, the same year that the EPA was established. NAAQSs are divided into primary standards based on risks to public health, especially of groups at risk such as infants, the elderly, and asthmatics, and secondary standards, based on public welfare such as visual range, materials degradation, and damage to crops or ecosystems. Regions where the primary standards of the NAAQS’s are met are referred to as attainment areas while those out of compliance are designated nonattainment areas.

In 1977, the CAA was again amended, primarily to mandate reductions of emissions from automobiles. The standards, especially for NO_x, were relaxed to give manufacturers more time to comply with the new requirements. 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 km⁻¹) until 1981. States are required to monitor criteria pollutant concentrations to demonstrate compliance, but the secondary effects of several these gases are also pose health and welfare concerns. Few locales violate the standards for CO or NO₂, but NO_x causes harm at concentrations well below the primary standard of 53 ppb (need a citation here). If a city had an annual average NO₂ level anywhere near the NAAQS for NO₂, it would risk severe photochemical smog – the summertime efficiency for ozone production ranges from 4 to 10 ppb O₃ per ppb NO_x.

The focus on compliance monitoring for NO₂ ignores the other, equally important members of the NO_y family such as HNO₃ that deposits quickly onto the Earth’s surface. As reported in the

1 recent ISA, “Overall, we conclude that there is a *causal* relationship between current levels of N
2 and S deposition and numerous biologically adverse effects on ecosystems across the United
3 States” (<http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=181712>). Conversion of the
4 existing network of NO_x monitors to NO_y monitors with a detection limit of 0.1 ppb would still
5 demonstrate compliance with the NO₂ standard but greatly increase the utility of the
6 measurements for model evaluation as well as for understanding nitrate deposition and formation
7 of photochemical smog, and haze.

8
9 The 1977 CAA Amendments also established permitting regulations and the first measures to
10 protect the stratospheric ozone layer. A formal permitting program is required for the Prevention
11 of Significant Deterioration (PSD) of air quality in attainment areas. Pristine areas such as
12 national parks and wilderness areas a designated Class I regions where no new sources of
13 pollution are allowed. Class II regions require new sources to meet stringent requirements and
14 Class III regions are allowed growth in transportation and industrialization as long as the
15 NAAQSS are met. Chlorofluorocarbon (CFC) regulation began with this Amendment, and the
16 ban became international through the Montreal Protocol, signed in 1987. Neither action
17 regulated emissions of nitrous oxide, the dominant source of stratospheric NO and responsible
18 for most ozone layer destruction.

19
20 Air pollution, especially ozone and PM, continued to be a problem in many American cities and
21 the CAA was again amended in 1990. The Nr-relevant aspects were aimed at controlling urban
22 smog and acid deposition. States were required to develop emissions inventories for reactive
23 organic compounds, carbon monoxide, and oxides of nitrogen (NO_x), but not NH₃ or N₂O. Over
24 the US, sulfate and nitrate are responsible for about 2/3 and 1/3 respectively of the direct
25 deposition of acids. The CAA Amendment of 1990 required emissions decreases of 10 million
26 tons of SO₂ and 2 million tons of NO_x relative to 1980 levels. Ammonia and ammonium,
27 although they contribute to acidity after entering terrestrial ecosystems [*Galloway, et al.*, 2003;
28 *NRC*, 2003] were not regulated by this legislation.

29
30 The 1997 revision of the CAA changed the standards for ozone and particulate matter (see Table
31 3-24). The averaging time for ozone was increased from 1 hr to 8 hr and a standard for
32 respirable particles was implemented. These particles with an effective aerodynamic diameter of
33 2.5 μm or less appear to have a more direct effect on human health; they also scatter visible
34 radiation more effectively than do larger particles, resulting in a greater impact on visual range,
35 the earth’s radiative balance, and climate. A sizable fraction of the mass of PM_{2.5} is condensed
36 Nr.

37
38 In addition to the CAA, the Regional Haze Regulations aim to restore Class I areas to their
39 natural levels of atmospheric clarity. As stated in the PM_{2.5} CD “Regional haze can be
40 described as any perceivable change in visibility (light extinction, visual range, contrast, or
41 coloration) from that which would have existed under natural conditions and is caused
42 predominantly by a combination of many anthropogenic sources over a wide geographical area,”
43 (EPA 2004). States are required to be in compliance by the year 2064.

44
45 Ozone and particulate matter, the two most recalcitrant of the criteria pollutants, cover large
46 spatial scales. These secondary pollutants are not released at the tailpipe; rather they form in the

1 atmosphere. Violations are declared on urban scales, responsibility for their control was
 2 assigned to States, but the physics and chemistry of smog and haze are regional. In the eastern
 3 US, ozone episodes often cover several states and involve pollutants emitted in upwind states
 4 that do not themselves experience violations [Chen, et al., 2003; Husar, et al., 1977; Logan,
 5 1989; Moy, et al., 1994; Ryan, et al., 1998]. The 1990 amendments to the Clean Air Act
 6 established, in part as a response to this scaling problem, the Ozone Transport Assessment Group
 7 (OTAG) and the Ozone Transport Commission (OTC). These have jurisdiction extending from
 8 Washington, DC to Maine. Progress has been made on regional control of emissions; the NOx
 9 SIP call, implemented in 2003 and 2004, has led to measurable improvements in ambient ozone
 10 and nitrate levels [Gego, et al., 2007; Sickles and Shadwick, 2007]. Experiences with ozone and
 11 PM provide a useful demonstration of why it is necessary to develop an integrated approach to
 12 management of reactive nitrogen.

13
 14 **Table 3-24.** Federal primary ambient air quality standards that involve reactive nitrogen,
 15 effective January 2008. Secondary standards are currently identical to the primary standards.
 16

Pollutant	Federal Primary Standard (NAAQS)
Ozone (O₃) 1-hr average 8-hr average	0.12 ppmv 0.08 ppmv
Nitrogen Dioxide (NO₂) Annual average	0.053 ppmv (100 µg m ⁻³)
Particulate Matter, coarse (PM₁₀) Diameter ≤ 10 µm, 24-hr average Annual average	150 µg m ⁻³ 50 µg m ⁻³
Particulate Matter, fine (PM_{2.5}) Diameter ≤ 2.5 µm, 24-hr average Annual average	35 µg m ⁻³ 15 µg m ⁻³

17
 18 **3.4.5 Terrestrial Impacts of Reactive Nitrogen**
 19

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

25
 26 The biologically reactive forms of nitrogen include: 1) a wide variety of chemically oxidized
 27 inorganic compounds (NO_y) -- the most important of which are NO, NO₂, HNO₃, N₂O₅, HONO,
 28 and nitrate ion (NO₃⁻), 2) a few chemically reduced inorganic compounds (NH_x) – the most
 29 important of which are NH₃ and ammonium ion (NH₄⁺), and 3) a huge array of organic
 30 compounds (RNH) that include amino acids, nucleic acids, peptides, proteins, nucleic acids, and
 31 peroxy compounds such as peraceacetyl nitrates (PAN).
 32

1 The impacts of increasing Nr in terrestrial ecosystems include (Vitousek et al. 1997):

- 2 1. Increased global concentrations of nitrous oxide (N₂O)
- 3
- 4 2. increased local and regional concentrations of other oxides of nitrogen (including nitric
- 5 oxide, NO) that drive the formation of photochemical smog, and ozone;
- 6
- 7 3. Losses of soil nutrients such as calcium and potassium that are essential for long-term
- 8 soil fertility;
- 9
- 10 4. Accelerated losses of biological diversity, especially among plants adapted to low-
- 11 nitrogen soils, and subsequently, the animals and microbes that depend on these plants;
- 12
- 13 5. More leaching of Nr to aquatic systems via both groundwater and surface runoff – a
- 14 cascade effect
- 15

16 *3.4.5.1 Nr Saturation and Ecosystem Function*

17
18 There are limits to how much plant growth can be increased by nitrogen fertilization. At some
19 point, when the natural nitrogen deficiencies in an ecosystem are fully relieved, plant growth
20 becomes limited by availability of other resources such as phosphorus, calcium, or water and the
21 vegetation can no longer respond to further additions of Nr. In theory, when an ecosystem is
22 fully Nr-saturated and its soils, plants, and microbes cannot use or retain any more, all new Nr
23 deposits will be dispersed to streams, groundwater, and the atmosphere. Nr saturation has a
24 number of damaging consequences for the health and functioning of ecosystems. These impacts
25 first became apparent in Europe almost three decades ago when scientists observed significant
26 increases in nitrate concentrations in some lakes and streams and also extensive yellowing and
27 loss of needles in spruce and other conifer forests subjected to heavy Nr deposition. In soils,
28 most notably forest soils because of their natural low pH, as ammonium builds up it is converted
29 to nitrate by bacterial action, a process that releases hydrogen ions and contributes to soil
30 acidification. The buildup of nitrate enhances emissions of nitrous oxides from the soil and also
31 encourages leaching of highly water-soluble nitrate into streams or groundwater. As negatively
32 charged nitrate seeps away, positively charged alkaline minerals such as calcium, magnesium,
33 and potassium are carried along. Thus, soil fertility is decreased by greatly accelerating the loss
34 of calcium and other nutrients that are vital for plant growth. As calcium is depleted and the soil
35 acidified, aluminum ions are mobilized, eventually reaching toxic concentrations that can
36 damage tree roots or kill fish if the aluminum washes into streams (Vitousek et al. 1997).

37
38 Forests and wetlands vary substantially in their capacity to retain added nitrogen. Interacting
39 factors that are known to affect this capacity include soil texture,
40 degree of chemical weathering of soil, fire history, rate at which plant material accumulates, and
41 past human land use. However, we still lack a fundamental understanding of how and why
42 nitrogen-retention processes vary among ecosystems much less how they have changed and will
43 change with time (Vitousek et al. 1997).

44

1 *3.4.5.2 Effects of Excessive Nr on Forests*

2
3 Trees growing in soils replete with Nr but starved of calcium, magnesium, and potassium can
4 develop nutrient imbalances in their roots and leaves. This may reduce their photosynthetic rate
5 and efficiency, stunt their growth, and even increase tree deaths. General effects of excessive Nr
6 on forests include the following (Cowling, 1989, Cowling et al. 1990, Cowling et al. 2002,
7 Garner et al.1989, Woodman and Cowling1987):
8

- 9 1. Increased productivity of forests soils most of which are Nr-limited throughout the US,
10 Nr deficiency of forest soils has been most fully quantified for pine forests in 14
11 southeastern states
12
- 13 2. Acidification of forest soils leading to decreased availability of nutrient cations including
14 calcium, magnesium, and potassium and aluminum toxicity, established most clearly in the
15 eastern US and both central and northern Europe
16
- 17 3. Nr saturation of forest soils, presently occurring mainly in high-elevation forests of the
18 eastern US and southeastern Canada
19
- 20 4. Ozone-induced predisposition of forest trees to damage by fungal diseases and insect
21 pests, most clearly established in the case of root disease and bark beetles in the pine forests
22 of southern California
23
- 24 5. Ozone-induced inhibition of photosynthesis in both softwood and hardwood tree species
25 most clearly established in controlled exposure studies in both the US and Europe at ambient
26 concentrations of ozone above 60 ppb. Such concentrations occur frequently throughout the
27 eastern US and southeastern Canada
28
- 29 6. Ozone induced direct injury to foliage, most clearly established in the case of “emergence
30 tip burn” in eastern white pine
31
- 32 7. Acidification induced decrease in frost hardiness of high-elevation conifer forests, most
33 clearly established in the case of red spruce in the northeastern US
34
- 35 8. Acidification induced alteration of beneficial symbiotic relationships in forest soils,
36 especially mycorrhizae, most clearly established in both northern and central Europe
37
- 38 9. Biodiversity losses in natural grasslands and forest areas caused by Nr induced decreases
39 in abundance of Nr-limited tree and grass species and replacement by Nr-loving weed
40 species, most clearly established in both Minnesota and California, and even more vividly in
41 The Netherlands
42
- 43 10. Decreases in visibility and increased haziness of the atmosphere at scenic vistas in
44 national and state parks and wilderness areas
45

46 *3.4.5.3 Decreases in Biodiversity*

1
2 In North America, dramatic reductions in biodiversity have been created by fertilization of
3 grasslands in Minnesota and California. In England nitrogen fertilizers applied to experimental
4 grasslands have led to similarly increased dominance by a few nitrogen-responsive grasses and
5 loss of many other plant species. In formerly species-rich heathlands across Western Europe, Nr
6 deposition has been blamed for great losses of biodiversity in recent decades, with shallow soils
7 containing few alkaline minerals to buffer acidification (Vitousek et al. 1997).

8
9 Losses of biodiversity driven by Nr deposition can in turn affect other ecological processes.
10 Experiments in Minnesota grasslands showed that in ecosystems made species-poor by
11 fertilization, plant productivity was much less stable in the face of a major drought. Even in non-
12 drought years, the normal vagaries of climate produced much more year-to-year variation in the
13 productivity of species-poor grassland plots than in more diverse plots (Vitousek et al. 1997).

14
15 *3.4.5.4 Nr Deposition (oxidized and reduced inorganic and organic compounds)*

16
17 As noted in section 3.3.1.1. of this report, atmospheric deposition of nitrogen oxides – and
18 chemically reduced forms of nitrogen (especially ammonia and ammonium ion), as well as many
19 different organic forms of Nr -- are causing a wide variety of sometimes beneficial effects
20 (increased growth and productivity of forests, natural grasslands, and crops planted in nutrient
21 deficient soils) and also sometimes adverse effects on terrestrial and aquatic ecosystems in many
22 parts of our country.

23
24 *3.4.5.5 Reactive Nitrogen Critical Loads*

25
26 The concept of critical loads of nitrogen and sulfur has been widely used in Europe, and
27 increasingly in the United States to define levels of acidic deposition that have harmful effects on
28 terrestrial and aquatic ecosystems. In recent years, the New England Governors and Eastern
29 Canadian Premiers Acid Rain Action Plan has led to evaluations of critical loads to surface
30 waters and forests in that region. Those studies identified many waters and forestlands that met
31 or exceeded critical load capacity for combined sulfur and nitrogen deposition both in the New
32 England states as well in the eastern Canadian provinces. The plan set target reductions of 20 to
33 30% for nitrogen oxides by 2007 and a 50% reduction of sulfur dioxides by 2010. These targets
34 are aimed at reducing long-range transport of air pollutants, acid deposition, and nutrient
35 enrichment of marine waters in the region.

36
37 In May 2006, a Multi-Agency Critical Loads Workshop was held which led to the formation of a
38 Critical Loads Ad-Hoc Committee (CLAD) under the National Atmospheric Deposition Program
39 (NADP) to, among other goals, “Provide consistency in development and use of critical loads in
40 the US”. One outcome is a project undertaken by NESCAUM to “estimate critical loads of sulfur
41 and nitrogen in atmospheric deposition for areas where sufficient knowledge, data, and methods
42 exist” and “to demonstrate the use of critical loads "as a tool for assessing environmental policies
43 and programs and managing natural resources.”

44
45 A February 2007 workshop sponsored by EPA on “The Assessment of Health Science for the
46 Review of the National Ambient Air Quality Standards (NAAQS) for Nitrogen (NOX) and

1 Sulfur Oxides (SOX)” expansively reviewed both ecosystem as well as human health effects
2 towards revision of the NAAQS. In policy discussions at the workshop, it was asked if critical
3 loads assessments were an effective means of improving ecosystem management, and if the
4 science was understood well enough to use critical loads as a management tool, with the
5 conclusion that there was limited use of critical loads approaches for management in the United
6 States. Nevertheless, the Multi-Agency Workshop on Critical Loads was cited as an agenda-
7 setting effort to resolve some of the science and policy issues that could help advance critical
8 loads approaches in the U.S. As noted above, the ISA has released a draft report primarily
9 focused on human health criteria, but in the Review Draft “Annexes for the Integrated Science
10 Assessment for Oxides of Nitrogen – Health Criteria” (EPA, 2007a) ecosystem effects were
11 summarized as follows:
12

13 “In addition to the contribution to precipitation acidity, atmospheric nitrogen oxides have
14 ecological effects. Total loading by both and wet and dry deposition is the relevant metric for
15 considering ecosystem impacts. At low inputs, nitrogen deposition adds essential nutrients to
16 terrestrial ecosystems. Most temperate forests are nitrogen limited; thus the inputs stimulate
17 growth. Anthropogenic nitrogen may influence some plant species different and alter the
18 distribution of plant species. At high nitrogen loading, where nitrogen inputs exceed nutrient
19 requirements, deleterious effects including forest decline associated with ‘nitrogen saturation’
20 are seen. In aquatic ecosystems, however, nitrogen is may or may not be limiting, but in
21 brackish waters atmospheric deposition of anthropogenic nitrogen is suspected of contributing
22 to eutrophication of some coastal waters and lakes.”
23

24 **3.4.6 Reactive Nitrogen Indicators and Limits in Europe**

25
26 The European Union has undertaken broad measures to manage Nr. Tables 3-25, 3-26 and 3-27
27 summarize several different environmental impacts, currently used indicators, and whether there
28 are current limit values set by the United Nations Economic Commission for Europe (UNECE)
29 or European Union (EU). These tables identify the main links to the cascade of reactive nitrogen
30 in the environment, the relevance and link to Nr of the effect/pollutant, and existing agreements
31 in which the effect is currently addressed. In addition, some impacts are more relevant than
32 others in relation to societal importance and the connection to the nitrogen cascade. The
33 categorization on a scale of 1 (highest relevance) to 5 (unimportant) provides a first level
34 prioritization for future mitigation activity. One aspect of this global view of nitrogen impacts
35 and metrics that is evident is the mix of “classical”- and “service”-based categories, consistent
36 with the need for an integrated approach to the management of nitrogen.
37

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1 **Table 3-25:** Summary of the effects of excess reactive nitrogen on human health in relation to currently used
 2 metrics, the existence of regulatory values, and the link to the nitrogen cascade. The relevance and link to RN
 3 provide a prioritization for future international action to mitigate the effects of excess nitrogen. The last column
 4 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	SOMO35	Y	NO _x emission	3	CLRTAP ¹ ; EU CAFÉ ²
-- other photochemical oxidants	Org. NO ₃ , PAN conc (atm)	N	NO _x emissions	5	indirectly CLRTAP et al
-- fine particulate aerosol	PM ₁₀ , PM _{2.5} conc (atm)	Y	NO _x , NH ₃ em	1	CLRTAP; EU CAFE
-- direct toxicity of NO ₂	NO ₂ conc (atm)	Y	NO _x	2	WHO ³ ; CLRTAP; EU CAFE
Nitrate contamination of drinking water	NO ₃ conc (aq.)	Y	NO ₃ leaching	2	EU EFD ⁴ ,
Depletion of stratospheric ozone	NO _x , N ₂ O conc/flux (atm)	N	NO _x , N ₂ O	3	Montreal Protocol
Increase allergenic pollen production, and several parasitic and infectious human diseases		N		5	None
Blooms of toxic algae and decreased swimability of in-shore water bodies	Chlorophyll a NO ₃ (&P) conc (aq)	N	Runoff, Nr deposition	1	OSPAR ⁵ ; HELCOM ⁶ ; Barcelona Convention

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

7 ¹ Convention on Long-range Transboundary Air Pollution

8 ² Clean Air for Europe

9 ³ World Health Organization

10 ⁴ Essential Facilities Doctrine

11 ⁵ Convention for the Protection of the Marine Environment of the North-East Atlantic

12 ⁶ Helsinki Commission

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1 **Table 3-26:** Summary of the effects of excess nitrogen on ecosystems in relation to currently used metrics, the
 2 existence of regulatory values, and the link to the nitrogen cascade. The relevance and link to N provide a
 3 prioritization for future international action to mitigate the effects of excess nitrogen. The last column summarizes
 4 existing links to international regulations and conventions.

	Metrics	Regulated?	Link to Nr cascade	Relevance and link to Nr	Regulatory or political convention
Ozone damage to crops, forests, and natural ecosystems	AFstY (O ₃ flux), AOT40	Y	NO _x	2	CLRTAP; EU CAFE
Acidification effects on terrestrial ecosystems, ground waters, and aquatic ecosystems	Critical loads	Y	Nr deposition	2	CLRTAP; EU CAFE, WFD
Eutrophication of freshwaters, lakes (incl. biodiversity)	BOD, NO ₃ conc (aq)	Y	Runoff, Nr deposition	3	WFD
	Critical loads	N			
Eutrophication of coastal ecosystems inducing hypoxia (incl. biodiversity)	BOD, NO ₃ conc (aq)	Y	Runoff, Nr deposition	1	OSPAR; HELCOM; Barcelona Convention
	Critical loads	N			
Nitrogen saturation of soils (incl. effects on GHG balance)	Critical loads	Y	Nr deposition	1	CLRTAP; EU CAFE
Biodiversity impacts on terrestrial ecosystems (incl. Pests and diseases)	Critical loads, critical levels (NH ₃ , NO _x)	Y	Nr deposition	1	CLRTAP; EU CAFE ; CBD

5
 6 **Table 3-27:** Summary of the effects of excess N on other societal values in relation to currently used metrics, the
 7 existence of regulatory values, and the link to the nitrogen cascade. The relevance and link to N provide a
 8 prioritization for future international action to mitigate the effects of excess nitrogen. The last column summarizes
 9 existing links to international regulations and conventions.

	Metrics	Regulated?	Link to Nr cascade	Relevance	Regulatory or political convention
Odor problems associated with animal agriculture	NH ₃ conc (atm)	N	same sources as NH ₃ emission	5	
Effects on monuments and engineering materials	Acidity in prec., prec./T O ₃ , PM	Y	NO _x , NH ₃	3	CLRTAP
Regional hazes that decrease visibility at scenic vistas and airports	PM _{2.5} conc (atm)	N	NO _x , NH ₃	4	
Global climate warming induced by excess nitrogen	N ₂ O, conc/flux (atm)	N	N ₂ O	1	UNFCCC
Regional climate cooling induced by aerosol	PM _{2.5} conc (atm)	N	NO _x , NH ₃	1	UNFCCC

1 Where there is a limit and the relevance for the nitrogen cascade is high, then this might be the
2 limiting factor for reactive nitrogen production and its associated losses to the environment.
3 Some limits might be more relevant in specific areas and less relevant in others. For example
4 NO₂ concentrations relevant for human health are limited to 40 ppb in urban areas, limiting
5 industry and traffic, but are probably not an issue in remote areas with low population densities.
6 In these areas, however, loss of biodiversity might limit nitrogen deposition and therewith the
7 sources in the region. The only way to determine the extent that critical thresholds are limiting is
8 by overlaying them for different regions and determining by monitoring data or by model
9 exercises where and which threshold is the limiting factor, putting caps on losses from sources.
10 A pre-classification of regions might be useful, e.g. urban regions, remote regions, marine areas,
11 etc.

12

13 **3.4.7 Tradeoffs of Reactive Nitrogen Impacts**

14

15 Because nitrogen is such an abundant and widespread element, and reactive nitrogen such a
16 critical component of the Earth's biosphere, associated impacts are many and pervasive. In many
17 cases the impacts of reactive nitrogen involve tradeoffs, i.e. mitigating one type of impact may
18 exacerbate others. Four such categories of tradeoffs are ammonia release from concentrated feed
19 lot operations (CAFOs), human nutrition, nitrification-denitrification, and nitrogen-carbon
20 related impacts.

21

22 **Finding F3-18.** The Committee notes that the effective management of reactive nitrogen in the
23 environment must recognize the existence of tradeoffs across impact categories involving the
24 cycling of other elements, particularly carbon, and recommends that integrated strategies for
25 nitrogen management be developed in cognizance of these connections.

26

27 *3.4.7.1 Ammonia release from Concentrated Animal Feed Operations (CAFOs)*

28

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

35

36 Current EPA policy (EPA 2007b) discourages States from controlling ammonia emissions as part
37 of their plan for reducing PM_{2.5} concentrations. Ammonia is a substantial component of PM_{2.5}
38 in most polluted areas of the US at most times. While it is true that reducing ammonia emissions
39 might increase the acidity of aerosols and precipitation, the net effect of ammonia on aquatic and
40 terrestrial ecosystems is to increase acidity. After being deposited onto the Earth's surface,
41 ammonium is under most circumstances, quickly nitrified, increasing the acidity of soils and
42 waters. The Committee is unaware of any evidence neither that ammonia reduces the toxicity of
43 atmospheric aerosols nor that high concentrations of ammonia occur naturally over any
44 substantive area of the US. Lower ammonia emissions will lower PM_{2.5} concentrations. Such
45 reductions in PM_{2.5} concentrations have been linked to reductions in morbidity and mortality.

46

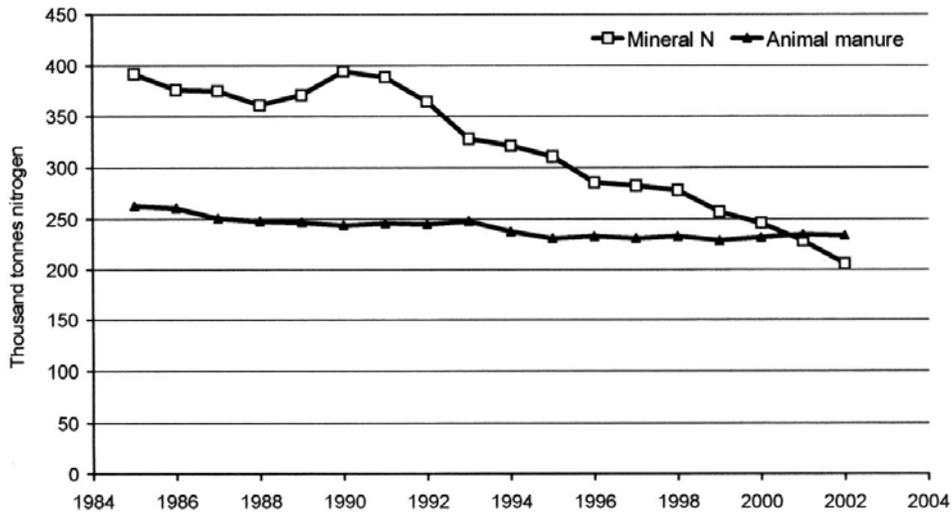
1 **Finding F3-19.** The Committee finds that the net benefit of ammonia emissions reductions
2 greatly outweigh any potential harm and recommends that the EPA presumption that ammonia
3 is not a PM_{2.5} precursor should be reversed and States should be encouraged to address
4 ammonia as a harmful PM_{2.5} precursor.
5

6 *3.4.7.2 Unintended Impacts of Lower Application Rates of Nitrogen for Crop Production*
7

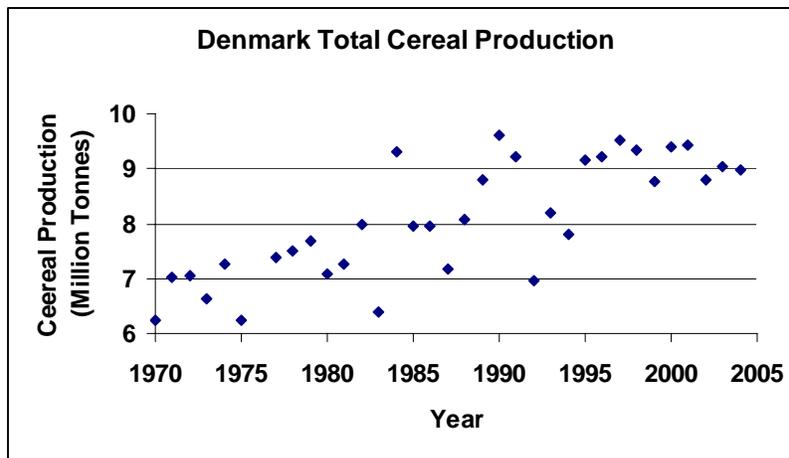
8 Crop production and environmental quality are potentially lost or gained at the expense of each
9 other. Although leakage of N from crop production systems cannot be eliminated, N losses can
10 be minimized substantially. One mechanism of decreasing leakage is to apply less N fertilizer to
11 croplands. For example Hu et al. (2007), using the SWAT model, predict that decreasing N
12 fertilizer application rates 10 to 50% in the upper Embarras River watershed in East central
13 Illinois, would decrease nitrate output to the river by 10 to 43%. This simple “solution” can
14 cause problems for crop production as yields and crop quality (protein content) may decrease,
15 causing economic loss to the farmer, decreased food quality for the consumer, and, at a global
16 scale, a reduction in food security.
17

18 Cropping systems managed in a consistent manner over time reach a functional equilibrium
19 between nitrogen inputs and outputs. Because crop yields are closely linked to the quantity of N
20 accumulation in above ground biomass at maturity (Cassman et al., 2002), there would be a
21 proportional decrease in crop yields in response to a decrease in the amount of N fertilizer
22 application. The magnitude of this yield reduction would depend on the magnitude of decrease
23 in the rate of applied nitrogen and the efficiency of nitrogen uptake from the applied nitrogen, as
24 well as interrelationships with other nutrients’ availability. Hence, yield reductions can be
25 mitigated, or even eliminated, if methods and fertilizer formulations used in fertilizer-N
26 application increased the efficiency of nitrogen uptake to offset the reduction in the amount of
27 applied nitrogen.
28

29 An example of the effect of decreasing N fertilizer input to cereal crop production on crop
30 production and crop quality as a result of national efforts to decrease N_r losses to the
31 environment from crop production is the situation in Denmark. In response to the European
32 Union Nitrate Directive synthetic fertilizer nitrogen use in Denmark was decreased (Fig. 3-19)
33 from approximately 400,000 mt (metric tons) in 1991 to 200,000 mt in 2002. Animal manure N
34 application decreased from 250,000 mt to approximately 240,000 mt during this time period.
35 Nevertheless, although N input into Danish cereal crop production decreased, cereal crop yield
36 remained relatively constant, as shown in Figure 3-20.



1 **Figure 3-19 Synthetic and livestock manure used as N fertilizer in Denmark (IFA**
 2 **2004).**

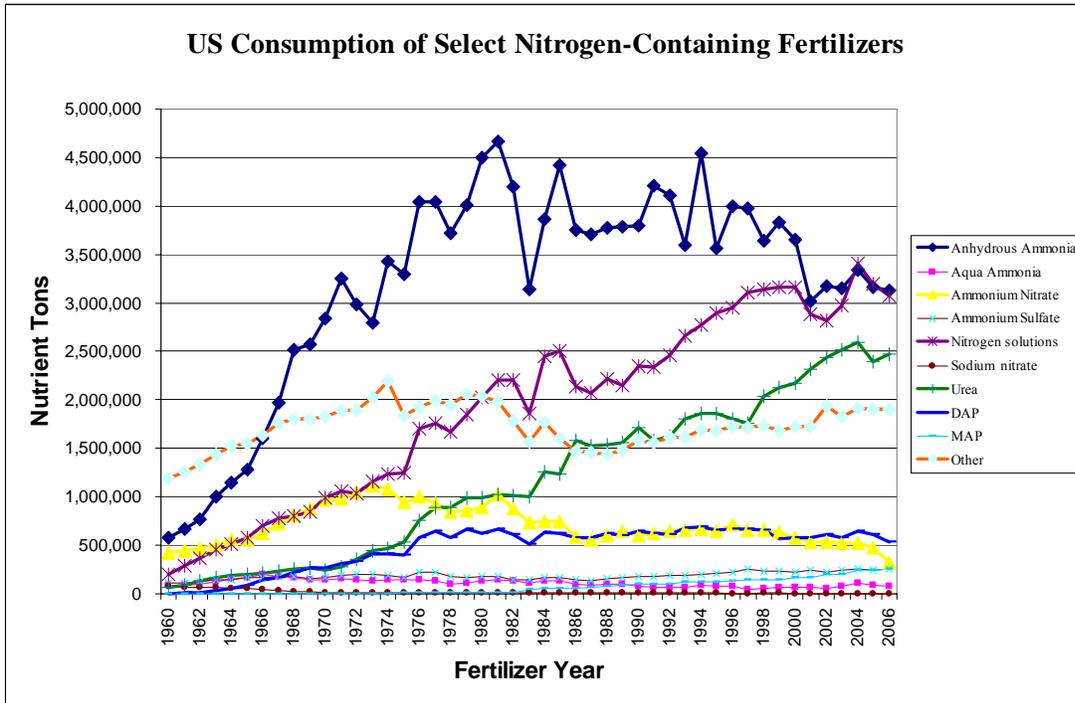


3 **Figure 3-20 Total cereal grain production in Denmark (FAOSTAT, 2007)**
 4

5
 6 If the methods used to apply N were to be modified to improve its overall efficiency, then it is
 7 possible to reduce N fertilizer inputs and maintain, or even increase crop yields depending on the
 8 magnitude of the improvement in NUE (see section 3.3). Although US fertilizer application has
 9 not declined over time, it has leveled off in recent years, as shown in Figure 3-21. Even so,
 10 yields, at least for corn grain, have continued to increase, a trend that has been in evidence since
 11 the mid 1970s, as shown in Figure 3-22.

12
 13 The effect of decreasing N fertilizer input can be estimated based on the current level of partial
 14 factor productivity (PFP) for applied N (average USA grain yield divided by the average N
 15 fertilizer application rate). Making the assumption that without a concerted effort to improve N
 16 fertilizer application methods, yields will decrease at 90% of the current PFP for N fertilizer
 17 (Cassman et al., 2003; Dobermann and Cassman, 2004). With a 10 to 50% decrease in N

1 fertilizer application the calibrated SWAT model predicted a 6 to 38% reduction in maize yield
 2 in the upper Embrarras River watershed (Hu et al. 2007).
 3



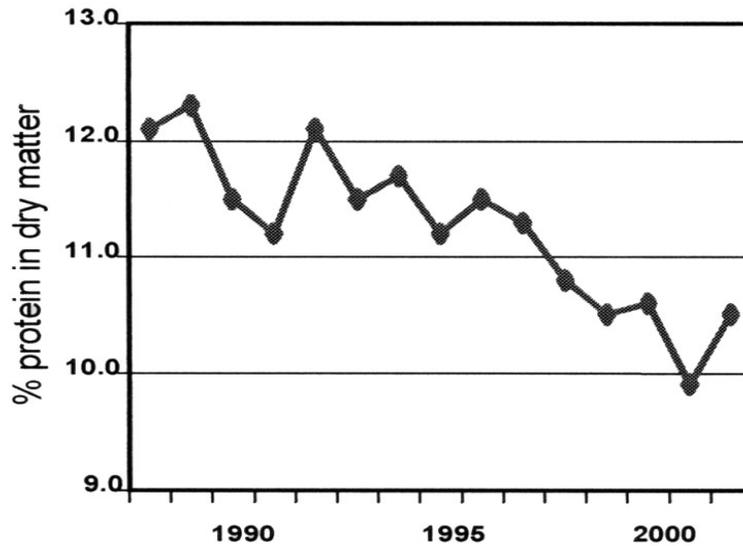
4 **Figure 3-21 Fertilizer application consumption in the US**



5 **Figure 3-22 Corn Grain Produced per Unit of Fertilizer N used in US (Fixen and Ford,**
 6 **2002)**

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 24 A negative impact of decreased Nr input into cereal crop production is the potential for a decline
 25 in grain quality as shown in Figure 3-23 as a decrease in grain protein content in Denmark.
 26 Grain protein content in wheat is critical for determining its quality for bread, for example in the
 27 USA a grain protein content of 12% is considered the threshold for good quality bread wheat,
 28 and nitrogen fertilizer application rate has a large influence on determining this trait (Cassman et

1 al., 1992). As can be seen, grain protein content has declined from 12 to 10% in Denmark over
2 the same period of lower fertilizer application rates.
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11 **Figure 3-23 Protein content of cereal grain in Denmark (IFA, 2004).**
12

13 Such trends raise several questions if declines continue or are found to be widespread. What is
14 the cost to the farmer and in terms of human nutrition and end-use value? Do these costs offset
15 the environmental benefits created by decreasing N flows from crop production areas? And, what
16 would be the regional and global impact if similar reductions in nitrogen fertilizer inputs to
17 agriculture were put in place in developed countries that represent the largest source of grain
18 exports to international markets? Finally, if protein yields are significantly reduced as a result of
19 lower N fertilization rates, more land may need to be brought into production. Because nearly all
20 prime agricultural land is already used for crop production, expansion of crop area will most
21 likely occur on more marginal land, such as the land currently in the Conservation Reserve
22 Program. Such conversion would result in additional N losses from these acres due to relatively
23 low nitrogen fertilizer efficiency that typically occurs on marginal land that has multiple soil
24 constraints to crop growth and yield.
25

26 *3.4.7.3 Unintended Impacts: Swapping Nitrogen between Environmental Systems*

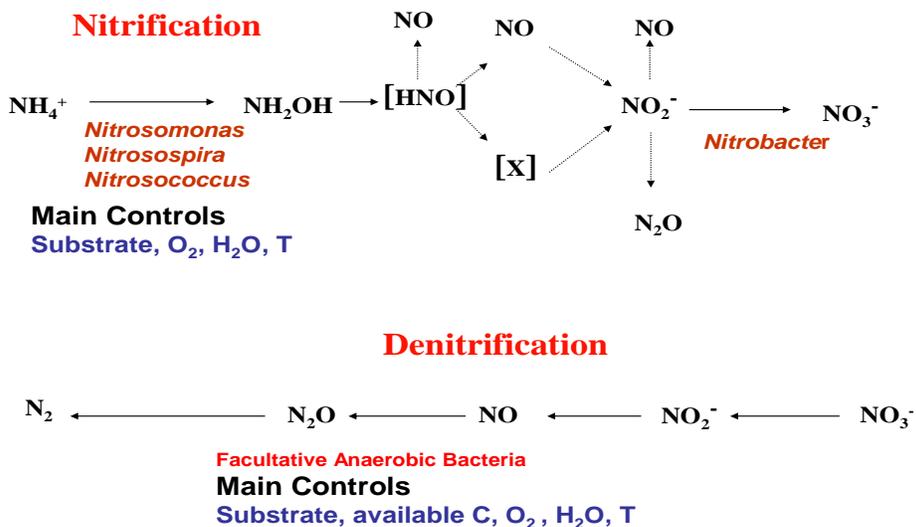
27

28 Nitrous oxide is produced in “natural” and agricultural soils, and all aquatic systems almost
29 exclusively as a result of microbial processes, nitrification and denitrification. As ammonium
30 ion is the initial mineral N product formed during organic matter mineralization and most
31 fertilizer used worldwide is ammonium based (e.g. urea, ammonium sulfate; FAO, 2007) the
32 suite of microbiological reactions that result in the release of gaseous N products need to be
33 considered.
34

1 Nitrification is the oxidation of ammonium ion to nitrate (Figure 3-24). Most commonly
 2 nitrification is a chemolithotropic process which consists of the conversion of ammonia to nitrite,
 3 which is then converted to nitrate by a second group of bacteria. The ammonia oxidizing
 4 bacteria (AOB) are obligate aerobes with some species that are tolerant of low oxygen
 5 environments. The most common genera of autotrophic ammonium oxidizers are *Nitrosospira*
 6 and *Nitrosomonas*, which result in the formation of nitrite. AOB are found in most aerobic
 7 environments where ammonia is available through the mineralization of organic matter or N
 8 compounds are added.

10 Biological denitrification is the dissimilatory reduction of nitrate and nitrite to produce NO, N₂O,
 11 and N₂ by a taxonomically diverse group of bacteria. These bacteria synthesize a series of
 12 reductases that enable them to utilize successively more reduced N oxides as electron acceptors
 13 in the absence of oxygen. The general reductive sequence is shown in Figure 3-24. In addition to
 14 the free living denitrifiers, symbiotically living Rhizobia in root nodules of legumes are able to
 15 denitrify nitrate and produce nitrous oxide (Mosier and Parkin, 2007).

17 The abundant denitrifiers are heterotrophs, which require sources of electron-reducing
 18 equivalents contained in available organic matter. Factors that most strongly influence
 19 denitrification are oxygen, nitrate concentration, pH, temperature, and organic carbon. The
 20 reductive enzymes are repressed by oxygen but not by ammonium. Nitrous oxide reductase
 21 appears to be more sensitive to oxygen than either nitrate or nitrite reductase. Therefore N₂
 22 production predominates in more anoxic sites and N₂O production may be greater in more
 23 aerobic conditions. However, the ratio of N₂ to N₂O emitted may also be affected by high nitrate
 24 concentrations and associated higher levels of electrical conductivity and osmotic stress and soil
 25 pH (low pH favors N₂O production).



26
 27 **Figure 3-24 Diagram of the nitrification and denitrification processes** (from Mosier and
 28 Parkin 2007)

1
2 Given these interactions among oxidized and reduced nitrogen species, it is important to
3 recognize the potential for unintended consequences to occur as a result of strategies aimed at
4 limiting one form of Nr in air or water that can lead to the increased production of other forms of
5 Nr. One such instance is the potential offsetting of the benefits of nitrate remediation at the
6 expense of increasing input of nitrous oxide to the atmosphere.

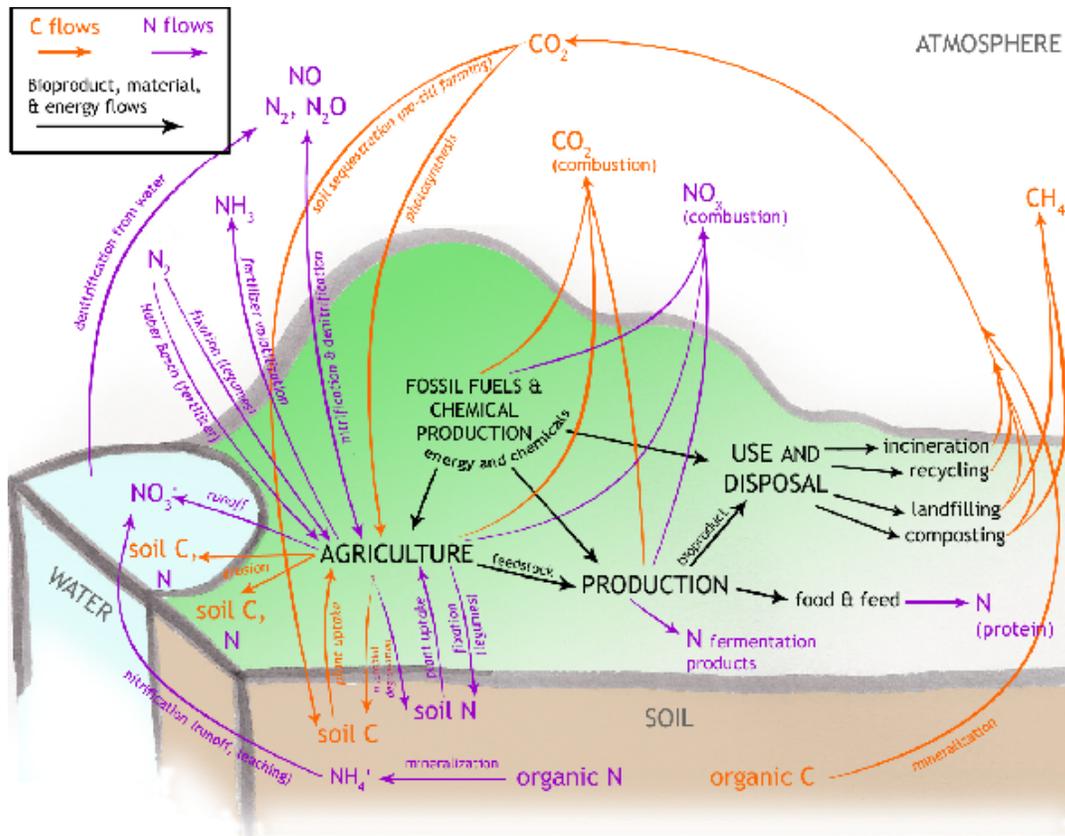
7
8 An example of such a situation involves nitrate leached from agricultural fields, much of which
9 could be removed from drainage water in natural or reconstructed wetlands. This process is ideal
10 if the denitrification process goes to completion, i.e. only dinitrogen gas is produced. If,
11 however, the process is incomplete, and NO and N₂O gases are emitted then the end result may
12 create a compensating risk that could be greater than that posed by the nitrate that is removed.
13 This is because NO continues to be reactive in the atmosphere and is eventually redeposited in
14 aquatic or terrestrial systems and N₂O is a greenhouse gas that has an atmospheric life time of
15 approximately 120 years and a radiative forcing of approximately 300 times that of CO₂ on a
16 hundred year time frame (IPCC 2001), and is a major source of NO in the stratosphere which
17 depletes stratospheric ozone (Crutzen 1981). If more of the nitrate denitrified is converted to
18 N₂O in wetlands than upstream or downstream, the environmental cost may be high. Hernandez
19 and Mitsch (2007) found that permanently flooded wetlands had lower N₂O/N₂ ratios of
20 emissions than did intermittently flooded wetlands. They also found that the ratio was higher in
21 the cold months even though the flux rates are much lower then. A full risk assessment needs to
22 be made to determine how much of such “pollutant swapping” is advisable.

23
24 A similar potential exists for Nr mediation in sewage treatment, for which current practice is to
25 convert ammonia/ammonium that mineralizes from excreted organic matter to nitrate through the
26 nitrification process. As nitrate containing effluent from sewage treatment flows into aquatic
27 systems the nitrate may be denitrified, resulting in N₂O production if denitrification is not
28 complete. The protein consumption by the ~301 million humans in the USA results in the
29 processing of ~ 2 Tg of N annually (~18.4 g N person⁻¹ d⁻¹), much of which flows through
30 sewage treatment facilities and ultimately leads to the production of 0.06 – 0.1 Tg of N₂O yr⁻¹ in
31 aquatic systems or soils to which sewage sludge is applied.

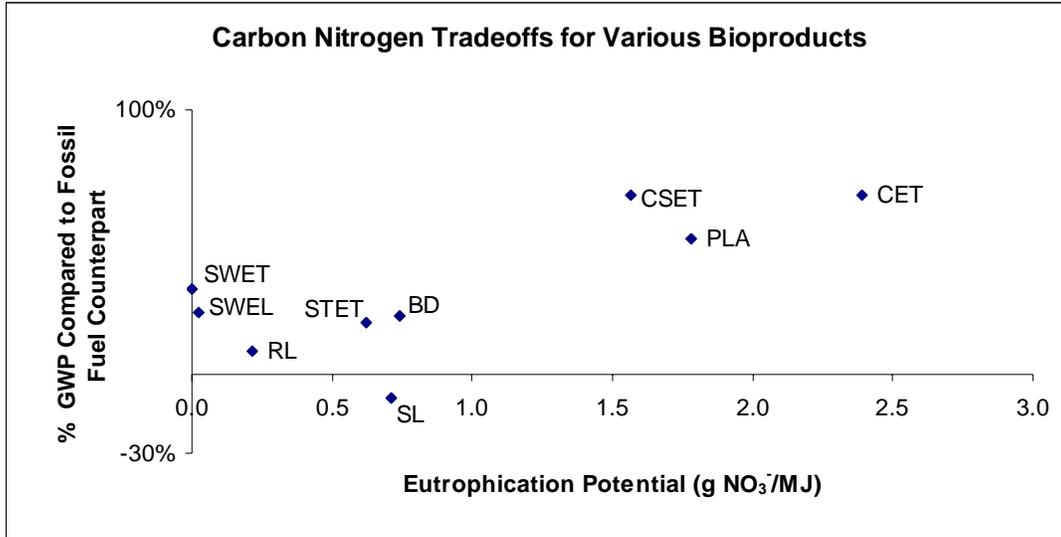
32 33 *3.4.7.4 Tradeoffs among Carbon and Nitrogen-Driven Impacts*

34
35 Many of the impacts on the environment to which reactive nitrogen contributes are also impacted
36 by other chemical species, notably carbon; there are several points of tangency between the
37 global carbon and nitrogen cycles, as depicted in Figure 3-25. The implication of these
38 interactions is that, in many instances, the perturbation of one cycle cannot be fully assessed
39 without including effects on the other. For example, proposals to develop bio-based products
40 (biofuels, but also other products) as the preferable alternative to fossil-based resources are not
41 impact-free. Such “trade-offs” may involve a single impact, e.g. global climate change for which
42 both carbonaceous gases and N₂O contribute, but may also involve trade-offs between impacts
43 that are not easily compared. Figure 3-26 shows the latter case in the form of global warming
44 impacts (for which carbon is a principal contributor) versus eutrophication impacts (for which
45 nitrogen is a principal contributor) for several different biofeedstock-product combinations
46 which are evaluated relative to the substituted commercial product made from fossil carbon. One

1 hundred percent would mean that the bio-based alternative is no better than the fossil-based
2 counter-product, while the negative region of the y-axis in Figure 3-26 represents net carbon
3 sequestration. It is difficult to make direct comparisons across disparate impact categories,
4 however Figure 3-26 suggests that, in choosing among alternatives, policies that aim to minimize
5 both sets of impacts would be preferred.
6
7
8



9
10 **Figure 3-25 Combined Carbon and Nitrogen Global Cycles (Miller et al. 2007).**
11



1
2 **Figure 3-26 Comparisons Between Global Warming and Eutrophication Impact Categories**
3 **for Various Bioproducts** (Miller et al. 2007). (Abbreviations: BD=Biodiesel; CET=Corn Ethanol;
4 CSET=Corn & Stover Ethanol; PLA=Polylactic Acid (Corn); RL=Rapeseed Lubricant; SL=Soybean Lubricant;
5 STET=Stover ethanol; SWEL=Switchgrass Electricity; SWET=Switchgrass Ethanol).
6
7

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Chapter 4: Integrated Risk Reduction Strategies for Reactive Nitrogen

4.1 Introduction

In Chapter 3, the environmental impacts and metrics associated with the emission of the various forms of reactive nitrogen were presented, and ways of organizing these into impact “categories” reviewed. As noted, reactive nitrogen has many impacts on the environment, impacts that are interrelated through the nitrogen cascade.

Finding F4-1. 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.

Once the nature and type of impacts are recognized, the environmental risk paradigm requires that these be translated into quantitative risk assessments, which are then 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 reactive nitrogen in the environment by the EPA follows an impact-by-impact approach which, with few exceptions, examines specific forms of nitrogen in either aquatic, atmospheric, or terrestrial systems. The principal regulatory authority pertaining to nitrogen is derived from two major legislative initiatives, the Clean Water Act (CWA) and the Clean Air Act (CAA).

4.2 Control Strategies for Reactive Nitrogen

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

1. Improved practices—in which the flux of reactive nitrogen 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, reactive nitrogen (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 reactive nitrogen introduced into the environment is lowered through preventive measures (e.g. precision fertilizer application, controls on NO_x generation)

- 1 4. Voluntary programs in which desired ends are achieved using private or government-
2 initiated agreements or through outreach and education.
3

4 **4.3.1 Command-and-Control**¹

5
6 Policy makers have traditionally used command-and-control strategies requiring individuals and
7 firms to meet mandatory guidelines. Such an approach evolved as the country was gearing up to
8 meet the requirements first established nationally through the CWA and CAA enabling
9 legislation in the 1970's. Because our capabilities to monitor contaminant concentrations and
10 predict environmental impacts was, generally, rudimentary, early emphasis was placed on
11 "technology-based" approaches for managing emissions, resulting in the promulgation of "best
12 practicable technology" controls, and eventually "best available technology" controls, the idea
13 being that mandating some level of control, even with uncertain improvements on impacts,
14 would be better, and less arbitrary, than other approaches of the time.
15

16 Nevertheless, both the CWA and the CAA had more specific goals that were aimed at protecting
17 human health, public welfare, and ecosystem health. For example the CAA required States to
18 develop Implementation Plans (SIPs) the approval of which depended on their ability, once
19 implemented, to meet ambient clean air standards. Likewise, the CWA required greater controls
20 to be implemented for certain water bodies for which technology-based limits alone were
21 insufficient to meet standards (this became the Total Maximum Daily Load program).
22

23 Over time, and as our abilities to monitor, predict, and understand impacts improved, it became
24 possible, or at least plausible, to tailor emission levels on a source-by-source basis, allowing the
25 firm in question to decide its own technological approach. Thus permits, which place strict limits
26 on the amount of pollution a firm is allowed to discharge over a specified period of time, have
27 become the main method for managing the majority of contaminants, including the various forms
28 of reactive nitrogen, in the environment.
29

30 **Finding F4-2 and Recommendation R4-1.** The INC finds that there is a need to regulate
31 certain reactive forms of nitrogen according to specific problems which have arisen in relation to
32 excess Nr, and concurs that enactment through command-and-control regulations is often the
33 most effective way to limit environmental impacts. However, for Nr we believe that a better
34 approach for an overall management strategy is the concept of defining acceptable total reactive
35 nitrogen critical loads to a given environmental system, and we therefore recommend that the
36 Agency work toward adopting this approach in the future. In carrying out this recommendation
37 the Committee recognizes that it will in many cases be necessary for the Agency to enter into
38 new types of research, policy, and regulatory agreements with other Federal, State, and Tribal
39 units based on cooperative, adaptive, and systemic approaches that derive from a common
40 understanding of the nitrogen cascade.
41

¹ Based on **Models in Environmental Regulatory Decision-Making**, National Research Council, 2007.

1 **4.3.2 Market Based Instruments for Pollution Control** ²

2
3 A fundamental shift in environmental management philosophy was initiated with the 1990 Clean
4 Air Act Amendments which combined regulatory requirements with market flexibility allowing
5 lower compliance costs through tradable permits. Most market-based policy instruments operate
6 on the principle that if the regulatory framework or some other factor sufficiently alters the
7 relative value of available decision choices for an individual or firm, subsequent decisions they
8 make will be in alignment with the policy maker's objective.

9
10 As an example, if a government wants to limit pollution in a river where a number of polluters
11 discharge it need not adopt a uniform command and control limit on each firm. Instead, a
12 regulatory cap on the total permissible pollution can be established at a lower pollution level and
13 permits to pollute that sum to that overall cap can be issued to all firms. Those firms having low
14 pollution control costs will have incentive to control more pollution than their permit allowance
15 and thus have permits they no longer need that can be sold to firms with high costs of pollution
16 control. Because the supply of permits (and the overall cap on the pollutant) is fixed, the
17 regulatory goal is achieved. The tradable permit thus brings about the desired reduction in
18 pollution level at lower cost than of the firms having high costs of pollution control were
19 required to control their full share and low cost of control firms were limited to their share of
20 control.

21
22 As with control strategies for Nr, there is no one universal market-based strategy that is
23 applicable to every policy maker's objective. For example, the nature of incentives available to
24 and effective with producers involved in over-fishing is different from landowners providing
25 environmental amenities. In the former case, the objective is to restrict the intensity of fishing. In
26 the latter case the objective is to encourage private landowners to provide environmental goods
27 and services and the lowest cost possible.

28
29 Evolution of new market-based strategies is a continuous process. Most strategies have been
30 customized over time to meet local needs. One can group such market based approaches under
31 the following conceptual headings:

- 32
33 1. Water Quality Tradable Permits: Every polluting entity is allowed to discharge pollutants
34 up to a certain pre-determined limit, defined in concordance with the terms of the CWA. The
35 entities discharging less than their allocated limit generate credits. Under this strategy,
36 credits can be traded with other polluting entities that have exceeded their allocated limit.
37
38 2. Auction Based Contracting: Environmental or conservation contracts are auctioned where
39 individual landowners place their bids to provide such goods or services from their land.
40 Two factors jointly determine the selection of the bids; the amount of the bid and the
41 expected value of the environmental or conservation benefit resulting from accepting the
42 bid.

² 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.

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1
2 3. Individual Transferable Quotas: An individual transferable quota (ITQ) is an allocation
3 privilege to extract a specified quantity of a resource among a selected number of quota
4 holders. The distinctive feature of the ITQ is that the privilege is transferable or leasable. An
5 ITQ may be a right to produce under favorable circumstances, such as a tobacco quota when
6 tobacco production would normally be limited.

7
8 4. Risk Indemnification for Specified Behavior: An example of this is crop insurance
9 designed to protect farmers from uncertainty in the adoption of best management practices
10 that provide a public good but are inherently riskier.

11
12 5. Easements: Conservation Easements or conservation servitudes refer to the case where a
13 land owner enters into a legally binding agreement to surrender certain property rights for a
14 specified period of time either voluntarily or for compensation. Such arrangements usually
15 provide public goods relative to the environment or conservation (see section 4.3.3).

16
17 The policy maker's objective, the local conditions and several other factors determine the
18 suitability of a particular market based strategy. For example, a tradable permit strategy is well
19 suited where offsets are possible. In the case of water quality it is not uncommon to find a
20 spectrum of polluters at different levels of contribution. A policy framework that facilitates the
21 emergence of multiple options for polluters to buy credits from more efficient controllers of
22 discharge or to invest in new equipment to achieve further reductions is likely to accomplish the
23 desired level of water quality at the least possible cost to the economy. Table 4-2 illustrates the
24 potential effective match-up of a number of market based approaches with specific situations.
25 Accompanying this chapter are two examples of the application of market-based approaches for
26 the design of water quality trading schemes for Nr in watersheds (Water Quality Trading to Meet
27 the Long Island Sound Wasteload Allocation in Connecticut and Water Quality Trading in the
28 Illinois River Basin).

29
30 Although there are significance differences between water and air quality trading, there are also
31 several potential barriers to effective trading systems for both media. These are related to
32 accountability and monitoring; establishing standards and management goals; complexities of
33 cross media and multiple source trading, including parity of sources; insurance that outcomes
34 would reduce risk (environmental benefit); economics and marketability of traded credits; and
35 transparency of the program including public outreach and stakeholder involvement.

1
2
3
4
5
6

Table 4-2 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

Auction Based Contracting

↑

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;

→

Tradable Permits

1 **Finding F4-3 and Recommendation R4-2 .** The committee finds that the costs of many of the
2 impacts of Nr emitted into the environment are external to the systems responsible for its
3 generation. Further, emissions trading within and across media will present opportunities and
4 challenges for mitigating Nr health and environmental impacts, and recommends that the Agency
5 pursue innovative trading strategies that are aimed at the cost effective control of major Nr
6 sources and chemical forms across media.

8 **4.3.3 Government programs, mandates, and policy conflicts**

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

17 The principal agricultural conservation programs in the US are administered by the USDA, and
18 consist of the Conservation Reserve and Wetland Reserve Programs (CRP and WRP, land
19 retirement programs), Environmental Quality Incentives. (EQIP, a “working lands” program),
20 various land preservation programs, and
21 technical assistance programs to

22 agricultural land managers (USDA also
23 manages price support programs and
24 insurance and disaster programs that,
25 collectively, have relatively little
26 potential for impacting Nr
27 management). Figure 4-1 illustrates
28 funding trends for major initiatives
29 showing the slowing of growth in
30 retirement and assistance programs
31 while preservation and EQIP have
32 increased more recently. The INC
33 committee is not able to provide
34 guidance on the appropriate levels of
35 funding for these programs. Slowing of
36 the CRP may be a result of energy
37 policy initiatives (see below). Increases
38 in EQIP appear to be associated with greater attention to livestock production, a trend that
39 reflects growing needs for better management practices in this area (see below and section 3.3).
40 Of concern to the committee is the need for more effective approaches aimed at encouraging
41 farmers and land managers to adopt proven conservation strategies at the field, farm, and feedlot
42 scale (e.g. more advanced testing methods, GPS-based variable rate fertilizer application,
43 conservation practices for conserving Nr), and landscape scale (e.g. riparian buffers and filter
44 strips, wetlands, and stream restoration). It is clear that the extent of such practices fall far below
45 the technological frontier.

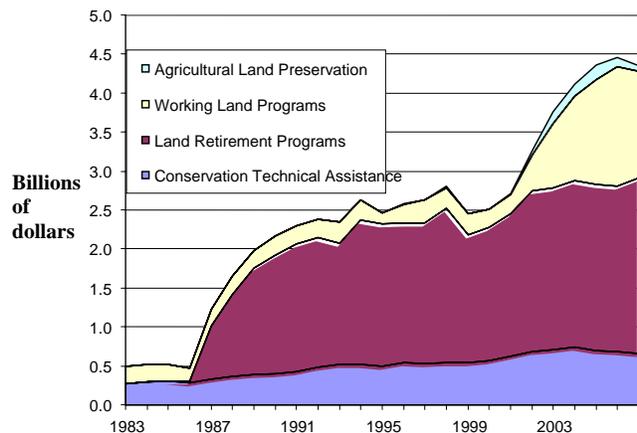


Figure 4-1. Trends in USDA Conservation Expenditures, 1983-2005

1 The construction and/or restoration of wetlands have received considerable attention in the past
2 two decades as a conservation method. Such an approach has several positive attributes including
3 promoting denitrification in watersheds containing or receiving Nr, flood protection, habitat
4 preservation, and recreational potential (Hey and Philippi, 1995). In the upper Mississippi basin
5 optimum siting of wetlands could result in as much as 0.4Tg of nitrate converted to N₂ (Hey,
6 2002; Mitsch et al., 1999). Of concern is the potential for the formation of N₂O in such systems
7 if not operated properly. Further details of wetlands as a management tool are presented as an
8 example in the accompanying text box.

9
10 **4.4 Risk Reduction Recommendations**

11
12 Human activities have significantly increased the introduction of Nr into the US environment,
13 and while there have been significant benefits resulting from food production, there have also
14 been, and continue to be, major risks to the health of both ecosystems and people due to the
15 introduction of Nr into the nitrogen cascade. To optimize the benefits of reactive nitrogen, and
16 to minimize its impacts, will require an integrated nitrogen management strategy that not only
17 involves EPA, but also coordination with other federal agencies, the States, the private sector,
18 and a strong public outreach program. The Committee understands that there are real economic
19 costs to the recommendations contained in this report. For each recommendation there will of
20 necessity be tradeoffs derived from the varying cost-effectiveness of different strategies.

21
22 In addition to specific recommendations given below, the Committee makes four over-arching
23 recommendations:

24
25 **OR 4-1.** We recommend that EPA pursue an integrated approach to develop the
26 understanding necessary for science-based policies, regulations, and incentives to avoid and
27 remediate the impacts of excess Nr on the environmental, human health, and climate. Such
28 integration must cut across media (air, land, and water), Nr form (oxidized and reduced),
29 federal agencies, and existing legislative statutes (e.g., EISA, the Clean Air Act, and the
30 Water Quality Act).

31
32 **OR 4-2.** We recommend that the EPA form an *Intra-agency Task Force on Managing Nr*
33 that builds upon existing Nr efforts within the Agency, with the main purpose being to
34 identify the most cost-effective approaches to avoid the negative impacts of Nr loads
35 cascading through the environment because it poses a significant threat to human health and
36 environmental quality, and is a powerful driver of climate change.

37
38 **OR 4-3.** We recommend that an *Inter-agency Task Force on Managing Nr* be formed, with
39 EPA as the lead agency that includes at a minimum USDA, DOE, DOT, NOAA, and USGS.
40 The responsibility of this Task Force is to coordinate federal programs that address Nr
41 concerns and help ensure clear leadership roles for specific functions in monitoring,
42 modeling, researching, and regulating Nr in the environment.

1 **OR 4-4.** We recommend that the purview of these task forces includes the following
2 elements
3

- 4 1) Research and science in support of integrated Nr management that includes
5 evaluation of critical loads; Nr budgets and life cycle accounting; monitoring as
6 the basis for informed policies, regulations, and incentive frameworks for
7 addressing excess Nr loads; the development and use of systemic models for Nr
8 management; new technologies; fertilizer and nutrient BMPs; development of Nr
9 indicators necessary for the assessment of effects related to excess Nr on human
10 health and the environment; combined C and N effects; indicators/endpoints,
11 costs, benefits and risks associated with the impairment of human health and
12 decline and restoration of ecosystem services). Research under this
13 recommendation should emphasize a systems approach.
14
- 15 2) The need for new regulations (and acceleration/extension of existing regulations)
16 and standards (e.g. a means of accomplishing the goals of CAIR, acceleration of
17 compliance of uncontrolled EGUs)
18
- 19 3) New indicators (e.g. combined NO_y/NH_x, ecological responses)
20
- 21 4) Education, outreach, and communication
22
- 23 5) Economic incentives, particularly those that integrate air, aquatic, and land
24 sources of Nr (markets, taxes, subsidies)
25
- 26 6) New infrastructures (e.g. stormwater control, treatment of Nr point sources)
27
- 28 7) Review of enabling legislation for purposes of extending regulatory authority or
29 streamlining procedures for enacting Nr risk reduction strategies.
30

31 Detailed management recommendations are as follows:
32

33 **Recommendation R4-3.** Improve detail and regularity of data acquisition for
34 fertilizer use by major crop (and for urban residential and recreational turf) and county
35 (or watershed) to better inform decision-making about policies and mitigation options
36 for nitrogen in these systems.
37

38 **Recommendation R4-4.** Improve estimates of N fertilizer uptake efficiency for the
39 major N-using crops and cropping systems based on direct measurements from a
40 representative range of production-scale farmer's fields to help guide prioritization of
41 risk mitigation strategies.
42

43 **Recommendation R4-5.** Improve loss estimates of NH₃, NO_x, nitrate leaching, and
44 N₂O; improve estimates of total Nr storage (or loss in soils coupled with organic
45 carbon); estimates of the fate of Nr goes (most losses are currently attributed to

1 denitrification in soils and water), and better identify the loss mechanisms from soil (e.g.
2 is denitrification the major loss mechanism or are loss estimates too large?)
3

4 **Recommendation R4-6.** Work closely with USDA and other agencies to identify
5 research and education priorities for prevention and mitigation of Nr applied to
6 agricultural systems.
7

8 **Recommendation R4-7.** Initiate research on the challenge of accelerating the rate of
9 gain in crop yields on existing farmland while substantially increasing N fertilizer uptake
10 efficiency.
11

12 **Recommendation R4-8.** Promote incentives for the use of advanced fertilizers and
13 enhanced efficiency products in crop and livestock agriculture.
14

15 **Recommendation R4-9.** Undertake an expanded research program for wetlands
16 design and management focused, in part, on Nr dynamics and removal.
17

18 **Recommendation R4-10.** Most NO_x sources can be controlled at the 90% level
19 (relative to uncontrolled combustion) with existing technology and at a reasonable cost,
20 and this should be an across the board goal. Accordingly, the Agency should continue to
21 reduce NO_x emissions from major point sources, such as EGU's, using a market
22 mechanism such as cap and trade. Under this scenario, it is likely that high efficiency,
23 low emission power plants will be built for energy needs. In implementing this approach,
24 the Agency should consider the mass of NO_x emitted per unit of power provided, rather
25 than past emissions, as the figure of merit for EGU's.
26

27 **Recommendation R4-11.** For total NO_x from all mobile sources the Agency should
28 consider a cap that decreases with time.
29

30 **Recommendation R4-12.** The Agency should implement and maintain programs such
31 as inspection and maintenance or road-side monitoring to ensure that most light-duty
32 vehicles meet emissions standards.
33

34 **Recommendation R4-13.** The Agency should require major sources of NO_x from
35 industrial fuel combustion to implement control technologies and/or include in cap and
36 trade programs.
37

38 **Recommendation R4-14.** The Agency should promulgate stricter NO_x emissions
39 standards for heavy duty diesel vehicles and off road vehicles including locomotives,
40 construction, farm, and landscaping equipment, and marine vehicles.
41

42 **Recommendation R4-15.** The Agency should encourage, through its sustainability
43 initiatives, replacement of EGUs powered by fossil fuels with cleaner energy sources
44 such as wind and solar.
45

1 **Recommendation R4-16.** Finally, the Agency should promote changes in lifestyle,
2 urban planning, and public transit conducive to energy conservation and reduced
3 emissions.

4
5 Through implementations of these recommendations the Committee believes that excess flows of
6 Nr into streams, rivers, and coastal systems be reduced by approximately 20% (~1 Tg) through
7 improved landscape management and without undue disruption to agricultural production. This
8 would include activities such as using wetland management (e.g., USDA Wetlands Protection
9 Program), improved tile-drainage systems and riparian buffers on crop land, and implementing
10 storm water and nonpoint source management practices (e.g., EPA permitting and funding
11 programs).

12
13 In addition, crop N-uptake efficiencies be increased by up to 25% over current levels through a
14 combination of knowledge-based practices and advances in fertilizer technology (such as
15 controlled release). Crop output can be increased while reducing total Nr by up to 20% of
16 applied artificial Nr, amounting to ~2.4 Tg below current levels of Nr additions to the
17 environment. These are appropriate targets with today's available technologies and further
18 progress is possible.

19
20 Acreage devoted to corn production has increased by approximately 10% for corn based ethanol
21 production during the past several years (with nearly one-third of the crop devoted to bioethanol
22 production), with fertilizer nitrogen increasing by at least 10% (**0.5 Tg N/yr**), largely to meet
23 biofuel feedstock crop demand. In the absence of Nr controls and a failure to implement best
24 practices, current biofuels policies will make it extremely difficult to reduce Nr losses to soils,
25 water and air (Simpson et al. 2008). Integrated management strategies will be required. In this
26 regard, the Committee endorses Section 204 of the 2007 Energy Independence and Security Act
27 (EISA) calling on the Agency to adopt a life cycle approach to the assessment of future
28 renewable fuel standards as a positive step toward a comprehensive analysis.

29
30 N₂O in the atmosphere is also increasing. For additional production of liquid biofuels beyond the
31 grandfathered amount in EISA, EPA has the power to exercise some controls on N₂O emissions
32 through the life cycle greenhouse gas accounting requirements. In addition, greenhouse gas
33 emissions trading will provide both opportunities and challenges with regard to mitigation of Nr
34 environmental and health impacts. The Committee therefore recommends that policies and
35 regulations that support implementation of emissions trading consider Nr impacts on GHG
36 emissions and reward reductions of N-related GHG. Biofuel subsidies should accurately account
37 for Nr contributions to GHG emissions, and individual biofuel plants should be certified for
38 GHG impact and serve as aggregators in the biofuel production life-cycle to reward reductions in
39 N₂O emissions through BMPs by farmers producing feedstock and use of co-products in
40 livestock diets.

41
42 There are two funding sources of significance authorized in the Clean Water Act that are used to
43 fund projects relevant to the control of Nr. Section 319 establishes state nonpoint source
44 management programs to plan for and implement management measures that abate sources of
45 nonpoint pollution from eight source categories, including both urban and agricultural sources.
46 Over the years section 319 has made available, through 60% matching funds, over \$1.6 billion in

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1 assistance. The much larger source of funding comes under Title VI of the CWA, which has
2 provided over \$24 billion (federal) for the construction of treatment facilities for point sources of
3 wastewater over the past twenty years, although only a fraction of this amount has been
4 dedicated to denitrification processes. These programs have been, and continue to be, important
5 ways of managing Nr in the urban environment.
6

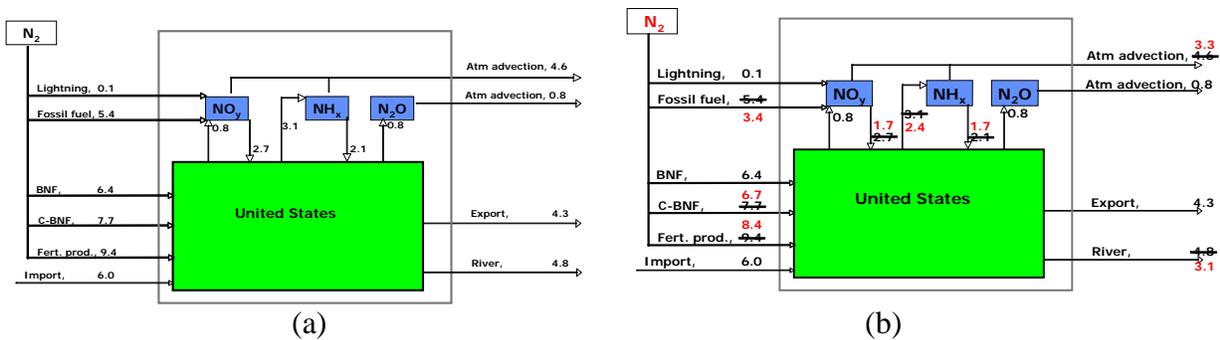
7 As shown in section 3.2, national loadings of Nr to the environment from public and private
8 wastewater point sources are relatively modest in comparison with global Nr releases, however
9 they can be important local sources with associated impacts. In most cases Nr ultimately finds its
10 way into municipal and private sewers and treatment systems where, irrespective of its initial
11 chemical form, it is partially or completely nitrified. Subsequent engineered complete
12 denitrification processes (including tertiary wastewater treatment, engineered or restored
13 wetlands, and algae production for biofuels) can convert the nitrate to N₂. Federal and State
14 assistance programs directed at construction of treatment plants are an important element Nr
15 control policy in the US. The committee recommends that a high priority be assigned to nutrient
16 management through a targeted construction grants program under the CWA. The committee
17 recommends that a high priority be assigned to nutrient management through a targeted
18 construction grants program under the CWA. The committee believes that **0.5 to 0.8 Tg N/yr** can
19 be cut from Nr inputs to the environment.
20

21 In spite of gains made over the last several decades in lowering the amount of NO_x emitted from
22 stationary and mobile combustion sources, the total amount of Nr released into the atmosphere
23 has remained relatively constant. This is related to the essentially unregulated release of
24 ammonia from livestock operations (due largely to increasing poultry and swine production),
25 which have expanded significantly, for example ammonia emissions from livestock production
26 has increased ~30% since 1970. The Committee suggests a goal of decreasing livestock-derived
27 ammonia emissions to approximately 80% of 1990 emissions a decrease of **0.5 Tg N/yr** (by a
28 combination of Best Management Practices and engineered solutions). This will reduce PM_{2.5}
29 by ~0.3 μg/m³ (2.5%); and improve health of ecosystems by achieving progress towards critical
30 load recommendations. Additionally we recommend decreasing ammonia emissions derived
31 from fertilizer applications by 20% (decrease by ~**0.2 Tg N/yr**), through the use of best
32 management practices.
33

34 The Clean Air Act (1970) and its Amendment (1990), have resulted in NO_x emissions that are
35 <50% of what they would have been without the controls. While this is an admirable
36 accomplishment, there is still a need to seek improvements, as NO_x emissions are still an order
37 of magnitude greater than at the beginning of the 20th century, and as a consequence there are
38 still negative impacts on both humans and ecosystems. Therefore, the Committee recommends
39 that the EPA expand its NO_x control efforts from the current reductions of emissions of
40 passenger cars and power plants to include other important unregulated mobile and stationary
41 sources. Notable NO_x emitters include heavy-duty on-road and all off-road mobile sources
42 (including rail and marine), as well as currently uncontrolled electricity generation and industrial
43 processes. Well-regulated electricity generating units and light duty vehicles currently eliminate
44 ~90% of the NO_x they would otherwise emit. Instituting 90% reductions for the major, currently
45 uncontrolled sources would reduce annual emissions by about **2 Tg N yr⁻¹**. This may be
46 sufficient to bring most of the US into compliance with the current O₃ NAAQS, but may still

1 leave some ecosystems with more Nr than the critical load. It is vitally important that the
 2 implementation of these controls not result in additional emissions of N₂O and NH₃ to the
 3 atmosphere.

4
 5 These recommendations, if implemented, would reduce Nr loadings to the environment in the
 6 US by approximately 25% below current levels. Figure 4.2 compares current and proposed Nr
 7 flows in the US. The Committee believes that these represent realistic intermediate targets based
 8 on current technology, however further reductions are needed for many N-sensitive ecosystems
 9 (e.g., estuarine and coastal waters). Developing these opportunities, and going beyond these
 10 recommended Nr reduction targets, are critical given the growing demand from population and
 11 economic growth for food- and fiber-production and energy use.
 12
 13



14
 15
 16 **Figure 4-2 (a)** Current levels of reactive nitrogen flows through the United States, and
 17 **(b)** reduced reactive nitrogen flows representing 25% reduction from current levels.
 18
 19

20 **Example: Water Quality Trading to Meet the Long Island Sound**
 21 **Wasteload Allocation in Connecticut**

22
 23 Pollutant trading is increasingly being promoted as a cost-effective means for attaining
 24 water quality standards. Connecticut and New York have been working with the EPA Long
 25 Island Sound Study (LISS) for more than 20 years to address low oxygen conditions (hypoxia) in
 26 Long Island Sound that have been linked to excessive loadings of nitrogen. A Total Maximum
 27 Daily Load (TMDL) for nitrogen, drafted by the two states and approved by the EPA in 2001, set
 28 a 58.5% nitrogen reduction target in 2014 from point and nonpoint source/stormwater (NPS/SW)
 29 sources. Connecticut has initiated a point source trading program for 79 municipal sewage
 30 treatment plants (STPs) to facilitate implementation of the TMDL wasteload allocation (WLA)
 31 and is investigating the potential for incorporating NPS/SW into the existing Nitrogen Credit
 32 Exchange (NCE).
 33

34 Several prerequisite conditions essential to the success of the current point source trading
 35 program have been met. Briefly, 1) all the STPs contribute to the same water quality problem; 2)
 36 the technology to remove nitrogen and meet the targets exists; 3) there are compelling member
 37 benefits to participate, especially cost savings; 4) sources can be easily monitored and tracked by
 38 end-of-pipe monitoring; 5) credit cost calculations are based on established and agreed upon

1 protocols founded in state legislation; 6) sources of nitrogen are diverse and create viable supply
 2 and demand conditions while reducing overall cost, with close control by a Nitrogen Credit
 3 Advisory Board (NCAB); and 7) transaction costs are low relative to credit prices. In operation
 4 since 2002, the NCE has proven to be a viable and effective mechanism for meeting the nitrogen
 5 WLA.

6
 7 The economic record of the NCE demonstrates the vigor of trading over the first five
 8 years of completed trades from 2002-2006 (Table 4-3). In sum, more than 10 million credits
 9 have been traded on the NCE representing more than \$22 million in economic activity.

Table 4-3 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

11 The use of geographically-based trading ratios is instrumental to the relative cost of meeting
 12 nitrogen reduction limits at the 79 treatment plants, which are scattered throughout the state (Figure
 13 4-2). Because nitrogen is non-conservative as it travels down rivers into the Sound, and the Sound's
 14 currents further affect relative impacts as they transport nitrogen, and the resulting algal blooms, to
 15 the hypoxic areas at varying efficiencies, location of each treatment plant makes a difference in
 16 relative impact on dissolved oxygen per pound of nitrogen discharged at end of pipe. Generally, the
 17 closer a POTW is to the edge of the Sound, and the closer to the hypoxic zone, the higher the
 18 trading ratio (Figure 4-3). For plants with high trading ratios, economics often favor treatment,
 19 while those with lower ratios may find the purchase of credits economically advantageous over
 20 treatment.

21
 22
 23 Further, there are economies
 24 of scale, i.e., larger POTWs (Figure
 25 4.2) often can remove nitrogen at a
 26 lower per pound cost than smaller
 27 facilities. Municipal Water Pollution
 28 Control Authorities (WPCA) can
 29 fully consider both location and cost
 30 when planning, and scheduling,
 31 potential nitrogen removal projects.
 32 The prices of nitrogen credits have
 33 been low (about \$1.65 - \$3.40 per
 34 equalized credit (Table 4-2)) but are
 35 expected to continue to increase over
 36 time as the lowest-cost projects are
 37 completed first and energy costs
 38 escalate. Because of the inevitable

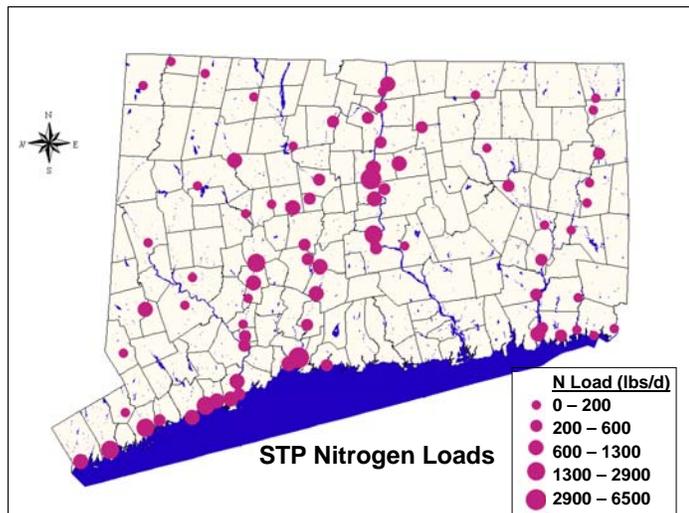


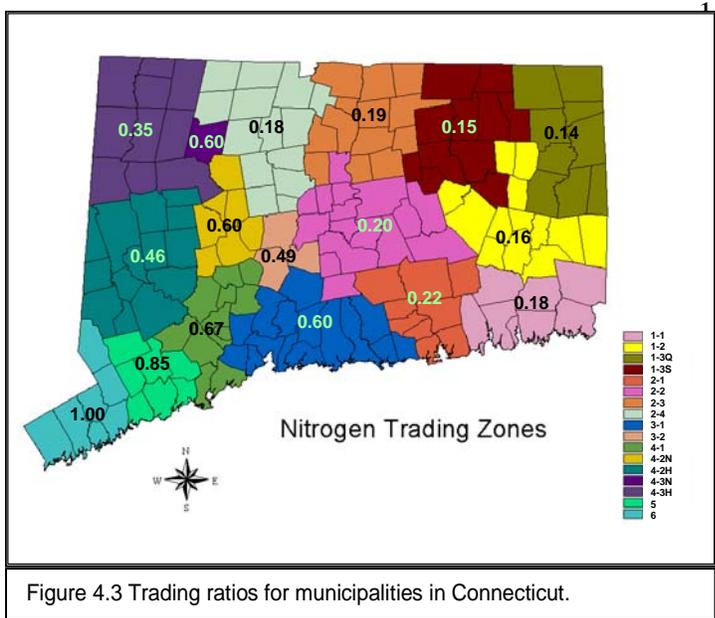
Figure 4.2. Relative nitrogen discharge (lbs/day) from 79 POTWs.

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1 increase in credit costs, decisions on treating vs. trading are re-evaluated on an annual basis by
2 WPCAs.

3
4 The point source NCE does not reflect a free market approach to trading. Demand is set
5 by the annual general permit limit and supply of credits is constrained by the availability of
6 Clean Water Fund dollars and the timing and location of nitrogen removal projects.
7 Nevertheless, there is a tendency towards implementing cost effective projects as STP authorities
8 decide whether it is less expensive to treat or buy credits, and try to predict when that break-even
9 point might occur that would warrant application for project funding.
10



Incorporating a nonpoint source/stormwater (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 nitrogen 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

29 transaction costs (or time spent negotiating the ground rules for NPS/SW trading) may be
30 considerably higher than for point source credits. Many of these obstacles can be overcome
31 by deferring to models and textbook costs and efficiencies for NPS/SW best management practices
32 (BMPs). Tracking will still be a challenge because of the sheer number and distribution of
33 BMPs that can be applied throughout the state that will have to be recorded and quantified.
34

35 Basic economic principles suggest that a free-market arrangement will not produce many
36 NPS/SW credits for market. Costs are much higher than for point source credits and a regulatory
37 approach must therefore be instituted to formalize the load allocation (LA) for nitrogen and to
38 structure participation by municipalities.
39

40 If a NPS/SW trading component were to be added in the future, it would most likely also
41 be an incentive-based program rather than a free-market approach. Nitrogen is difficult and
42 costly to control in Connecticut's urban/suburban setting, and reductions are unlikely to be cost
43 competitive with POTW credits in a free market system. However, because municipalities are
44 required to implement the Phase II stormwater permit, and various federal, state and local
45 programs that require or emphasize NPS/SW management, there may be benefits of an incentive-
46 based approach to offset some of those costs. For example, payment for NPS/SW reductions at

1 the same credit prices paid to POTWs under the NCE would help defray costs, and encourage
2 additional nitrogen reductions from stormwater/NPS sources. Connecticut and the NCAB will
3 continue to evaluate and explore the viability of these options.

Example: Water Quality Trading in the Illinois River Basin

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

20 Taking into account the economics of using wetlands to manage reactive nitrogen adds
21 yet another dimension to site selection. Based on the results of the Water Environment Research
22 Foundation's study (Hey et al., 2005), The Kinship Foundation sponsored a study (Kostel et al.,
23 in preparation) to define the market for producing and selling Nr (as NO₃) credits. For this
24 analysis, a real, potential market area was selected: the Illinois River watershed in Illinois—the
25 tributaries draining Wisconsin, Indiana and Michigan were excluded. The producers of nitrogen
26 credits were identified as “nutrient farmers” and they became the “sellers” of nitrogen credits.
27 The “buyers,” of nitrogen credits, were restricted to municipal and industrial wastewater
28 treatment facilities, those facilities that hold an NPDES permit. This restriction, of course,
29 resulted in a considerable understatement of the market size since the identified buyers emit less
30 than 11% of the total aquatic nitrogen load (David and Gentry, 2000), which finds its way to the
31 Mississippi River—air emission/deposition and agriculture account for the remaining 89%.

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

45 All 290 permitted dischargers (buyers) are geographically distributed as shown in Figure
46 4-4. The mass loading of the buyers (2,423 tons/month) is reflected in Figure 4.5. 89% of the

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1 demand comes from the northeastern corner of the basin (Upper Fox, Des Plaines, and
2 Chicago/Calumet sub-watersheds), the Chicago metropolitan area. As illustrated by Figure 4-6,
3 41% of the wetland restoration area (using the criteria discussed above) were identified in the
4 southwestern corner of the watershed (Lower Illinois, La Moine, Macoupin, Lower Sangamon,
5 and Middle Illinois sub-watersheds), where the floodplain is almost entirely leveed. For the
6 market study, the available load of reactive nitrogen (NO₃) by season and sub-watershed was
7 mapped as illustrated in Figure 4-5. The nitrogen load was computed using water quality and
8 flow data collected by the U. S. Geological Survey from 1987-1997. The wetland and
9 wastewater cost functions are described in Hey *et al.*, 2005; however, the wetland cost functions
10 were modified for the market study to reflect the variability of land costs across the watershed
11 (i.e., higher land values in urban Chicago vis-à-vis lower land cost in rural Illinois). This
12 variability is reflected in the spatial distribution marginal costs shown for the spring marginal
13 costs grafted in Figure 4-8. As previously noted, wetland treatment costs vary by time of year
14 because the level of microbial activity, which drives the denitrification process, varies with water
15 temperature. So, in the winter more wetland area is required than in the summer to treat an
16 equivalent load of reactive nitrogen.

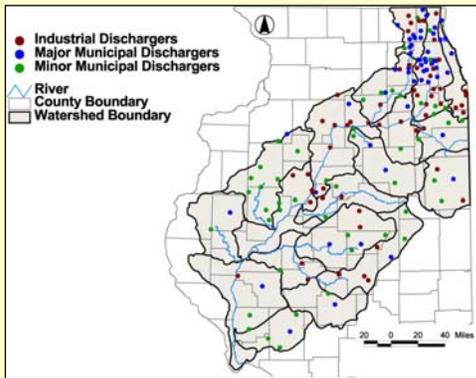


Figure 4-4: Distribution of municipal (> 1 MGD discharge), and industrial dischargers in the Illinois River Watershed; symbols may represent more than one discharger at that location.

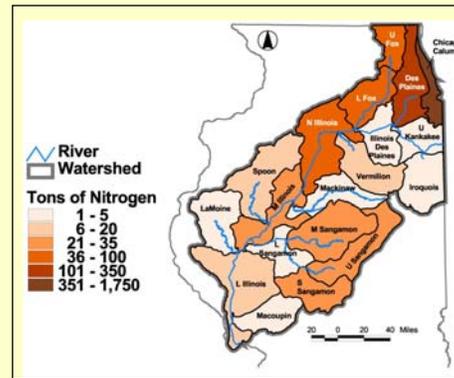


Figure 4-5: Distribution of total nitrogen emissions by sub-watershed.

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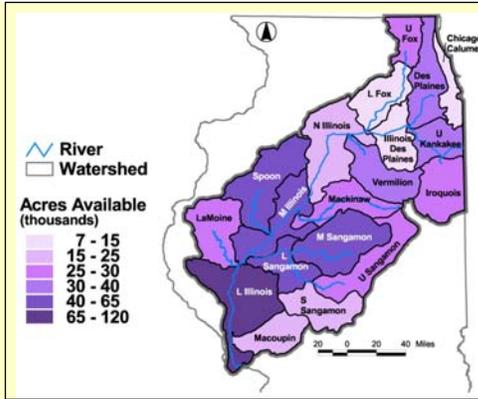


Figure 4-6: Potential land availability in the 100-year flood zone for nutrient farming in each sub-watershed in the Illinois River Watershed.

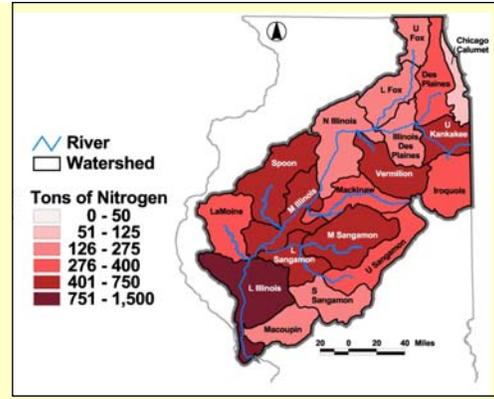


Figure 4-7: Spring available total nitrogen load by sub-watershed.

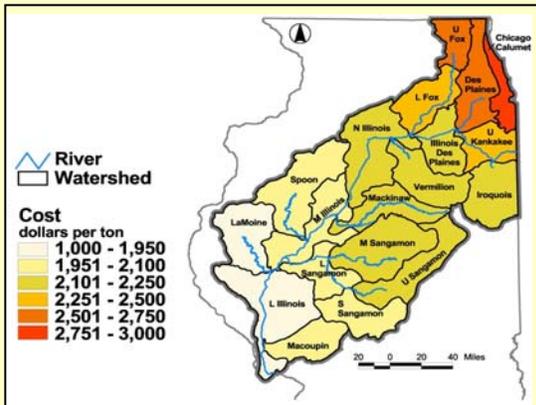


Figure 4-8: Spring marginal cost (price) by watershed.

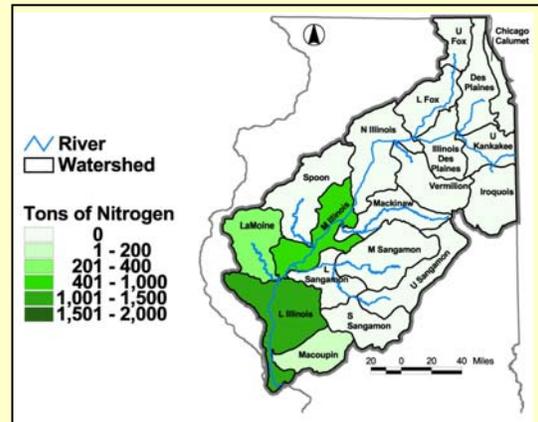


Figure 4-9: Unrestricted spring credit sales (tons/month) by sub-watershed

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 our analysis, we 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 4-9); 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

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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 nitrogen credits exceeding the mass of reactive nitrogen 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 4.4). 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 4.4: 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 ¹	\$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

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

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).

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