Nitrogen in Agriculture: Balancing the Cost of an Essential Resource

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Abstract

Nitrogen (N) is central to living systems, and its addition to agricultural cropping systems is an essential facet of modern crop management and one of the major reasons that crop production has kept pace with human population growth. The benefits of N added to cropping systems come, however, at well-documented environmental costs: Increased coastal hypoxia, atmospheric nitrous oxide (N_2O), reactive N gases in the troposphere, and N deposition onto forests and other natural areas are some of the consequences of our inability to keep fertilizer N from leaving cropped ecosystems via unmanaged pathways. The N cycle is complex, and solutions require a thorough understanding of both the biogeochemical pathways of N in agricultural systems and the consequences of different management practices. Despite the complexity of this challenge, however, a number of technologies are available today to reduce N loss. These include adding rotational complexity to cropping systems to improve N capture by crops, providing farmers with decision support tools for better predicting crop fertilizer N requirements, improving methods for optimizing fertilizer timing and placement, and developing watershed-level strategies to recapture N lost from fields. Solutions to the problem of agricultural N loss will require a portfolio approach in which different technologies are used in different combinations to address site-specific challenges. Solutions will also require incentives that promote their adoption.

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INTRODUCTION

The centrality of nitrogen (N) to living systems is well recognized: N is a primary constituent of nucleotides and proteins and thus is essential for life. Moreover, biologically available N is in short supply in most terrestrial ecosystems; at least as much as any other nutrient, the availability of N limits plant growth and primary production, fundamentally affecting the structure and function of most ecosystems.

Nowhere is N more important than in agricultural systems; the addition of N to sustain and increase crop yields is a pervasive and fundamental feature of modern crop management. Added N allows farmers to simplify the plant community by displacing the need for N-fixing plants; it alters microbial community structure in ways that affect decomposition and other ecosystem processes such as nitrification, denitrification, and methane (CH_4) oxidation; and it alters the palatability of crops to pests, thereby affecting trophic relationships. The net benefits to humans are huge—substantially more food can be grown on a given area of land—thereby increasing its human carrying capacity and simultaneously relieving some of the pressure for new land clearing with its concomitant costs. This is the abundant harvest of intensive N management: improved human health and well-being.

On the flip side, there are significant costs associated with agricultural N additions: N is mobile, hard to contain, and even N that is efficiently conserved and taken away in crop harvest eventually makes its way back to the environment. Most of the N mobilized from agricultural systems is reactive; it is present in forms that are biologically active in soils and surface waters and/or chemically reactive in the atmosphere. Agriculturally based increases in reactive N are substantial and widespread, and they lead to losses of biological diversity, compromised air and water quality, and threats to human health across large areas of Earth.

By far the largest pool of N in the biosphere is atmospheric N_2, which makes up 79% of dry air. However, the triple bond that binds the N atoms in N_2 is exceptionally strong, breakable by only a few processes. In nature, the energy required to break this bond can be provided by lightning or, much more commonly, by

N_2-fixing microbes with specialized enzymes capable of converting N_2 to NH_3. Industrially, this bond is broken with heat and pressure, and on land more N_2 is now fixed by industrial than by biological pathways.

Most of the N_2 fixed by industry occurs during N fertilizer production via the Haber-Bosch process, in which natural gas (CH_4) is burned to produce hydrogen that is then reacted with N_2 under high temperature and very high pressure to form ammonia (NH_3). The discovery of this process early in the last century and its commercialization post-World War II have provided a substantial and growing fraction of the world's agricultural N needs (**Figure 1**). This process also provides a majority of the new reactive N that now affects nonagricultural environments.

The aims of agricultural N management are to provide enough N to plants to maximize growth and subsequent crop yields and also to keep it out of other ecosystems, particularly those for which added N is harmful. The ultimate source of fixed N, whether biological or industrial, is unimportant; ecosystems receiving N respond similarly because organisms cannot differentiate among different sources. However, the form in which N is delivered is vitally important: Different forms of N (**Table 1**) are available at different rates to different organisms, and some forms are more readily lost by various hydrologic and gaseous pathways (**Figure 2**). Moreover, the extraordinary value of fixed N to autotrophs and to many heterotrophs ensures that it follows a complex and sometimes circuitous biogeochemical path through most ecosystems.

Here we describe the basis and history of N use in agriculture and the increasingly costly environmental price paid for this use. We then describe the basic biological processes that transform N as a prelude to discussing the fundamental challenge of how to use N without causing harm. We consider strategies that can address this challenge, i.e., technological fixes that improve N management. We conclude with the suggestion that a portfolio approach including both biophysical and sociological

Table 1 Main forms of nitrogen (N) in the environment and their oxidation states (43)

Name[a]	Species	Oxidation State
Organic N	R_{NH_3}	−3
Ammonium	NH_4^+	−3
Ammonia (g)	NH_3	−3
Dinitrogen (g)	N_2	0
Nitrous oxide (g)	N_2O	+1
Nitric oxide (g)	NO	+2
Nitrite	NO_2^-	+3
Nitrogen dioxide (g)	NO_2	+4
Nitrate	NO_3^-	+5

[a](g) indicates gas at STP.

measures is necessary to effectively manage this critical resource.

NITROGEN IN AGRICULTURE

The N cycle is complex (**Figure 2**) in agricultural as well as in other ecosystems. Its complexity is both boon and bane: While on the one hand, complexity provides multiple points of management intervention; on the other hand, it hides interactions among different processes and proscribes single, easy solutions to management challenges. At the outset, understanding N in agriculture requires understanding the fundamental need to match N supply to crop N requirements and the means by which we have achieved this goal in the past.

Nitrogen Balances of Cropping Systems

In any cropping system, N removed intentionally in crop yield and unintentionally by other means must be replaced if productivity is to be sustained. This is true for all plant nutrients, of course, including phosphorus, potassium, and calcium and those elements required in lesser amounts, such as magnesium and boron. What makes N different is the absence of a mineral-bound weatherable N pool in most soils. Unlike other elements, there is no potentially available N in the rocks from which most soil is derived, and new N must come from outside the
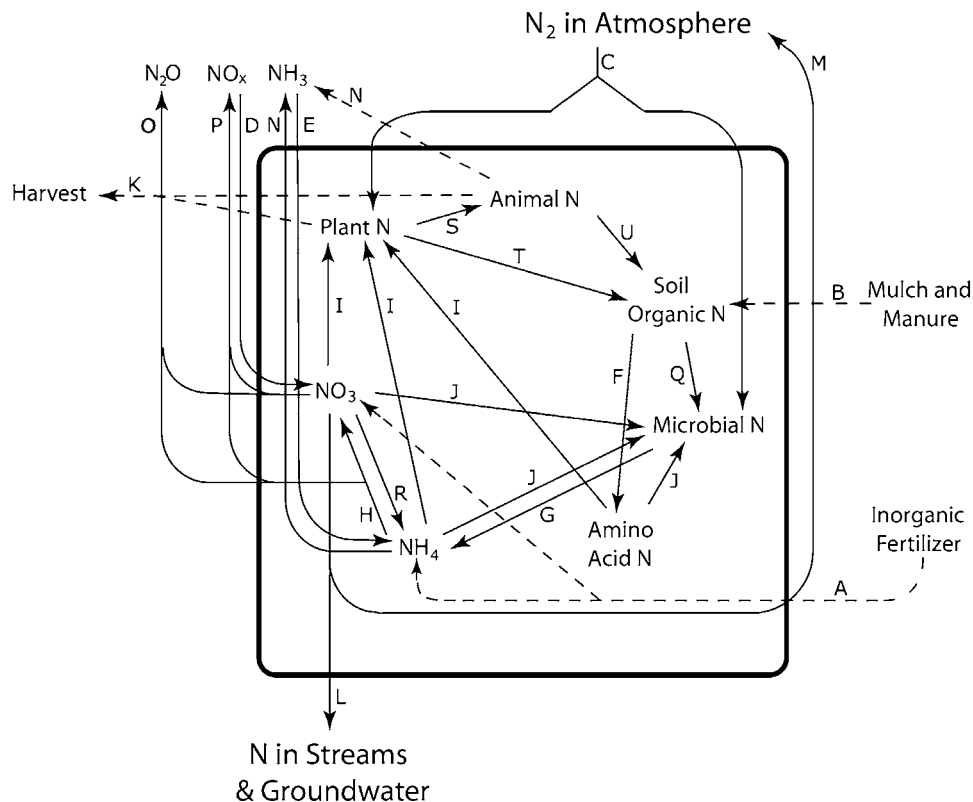
Figure 2

Pathways of nitrogen (N) cycling in agricultural ecosystems. The N cycle is complex, with multiple transformations and oxidation/reduction reactions; indeed, this depiction is a substantial simplification of the N cycle. Transformations of N shown in solid lines occur in all ecosystems; those shown with dashed lines are particular to (or particularly important within) agricultural systems. Major fluxes of N shown here include, A, additions of industrial fertilizer, typically in inorganic forms or forms like urea that are rapidly converted to inorganic forms in soil; B, additions of organic N in manure and mulches; C, biological N_2, fixation by microbes symbiotically associated with plants and by free-living microorganisms; D, atmospheric deposition of reactive N in oxidized forms; E, atmospheric deposition of ammonia (NH_3) and ammonium; F, mineralization of organic N via mobilization of amino acids through the action of extracellular enzymes; G, mineralization of organic N via release of ammonium by microbes; H, nitrification of ammonium, ultimately to nitrate; I, plant uptake of biologically available N; J, microbial immobilization—the uptake of biologically available N by microbes; K, losses of N in harvested products; L, losses of N in solution to streamwater and groundwater; M, denitrification to dinitrogen; N, NH_3 volatilization from both fields and from intensive animal production systems; O, losses of nitrous oxide (N_2O) produced during nitrification and denitrification; P, losses of reactive oxidized N produced during nitrification and denitrification; Q, uptake of organic N by microbes during decomposition; R, dissimilatory reduction of nitrate to ammonium; S, consumption of plant N by animals; T, flux of N to soil in plant litter; and U, flux of N to soil from excretion or animal death.

plant-soil system. Some N is added annually from rainwater and from dry deposition onto leaf and soil surfaces, but except in highly polluted regions, the majority ultimately must come from the fixation of atmospheric N_2.

In ecosystems without a substantial annual loss of N via harvest, soil N stores equilibrate at some stable level: N that is released from decomposing organic matter gets taken up by plants and then is more or less replenished by

Table 2 Nitrogen (N) removal rates in representative grain crops (82)

Crop	Yield MT ha^{-1}	Percent N	Grain N removed kg ha^{-1}
Corn (*Zea mays*)	10.0	2.6	260
Wheat (*Triticum aestivum*)	5.4	2.0	108
Rice (*Oryza sativa*)	7.9	1.8	142
Sorghum (*Sorghum bicolor*)	9.0	3.0	270
Soybean (*Glycine max*)	2.8	6.3	176

the return of organic N in roots, stems, and leaves to soil. However, the removal of N in crop yield means that less N enters the soil as plant residue than is released in decomposition, and over time there will be less N recycled for future plant use. Under intensive cropping, N depletion can happen quickly. In the early 1900s, for example, corn (*Zea mays*) grain yields on U.S. farms averaged 1.6 tonne (MT) hectare (ha)$^{-1}$ [25 bushel (bu) acre^{-1}]. At a grain N content of 2.6%, this effectively removed about 42 kg N ha^{-1} year^{-1} from soil N pools. Many arable soils contained total N stores of 3–15 MT N ha^{-1} prior to cultivation, resulting in a soil N depletion rate as high as 1% year^{-1} in net removal by harvested products alone. These extraction rates help to explain why soil N pools were depleted substantially following only 30–40 years of cropping (1, 2). Today's more common yields of 10 MT ha^{-1} (160 bu acre^{-1}) and higher would make dependency on N from stored soil organic matter (SOM) even less sustainable.

Other pathways of N loss exacerbate this imbalance in cropping systems. Annual crops are active only part of each year, and consequently other vectors of N loss, both hydrologic and gaseous, can be at least as important as harvest removals. The net result is that biologically available N is used inefficiently in annual crop systems. N balance studies from the 1930s show that 50% or less of fertilizer N applied to corn typically makes it into the crop (3), a fraction that has not improved substantially in 50 years of on-farm measurements (4).

Managing N well—ensuring both an adequate and an efficient supply for those plants unable to fix atmospheric N$_2$—thus becomes a key piece of the agricultural sustainability puzzle. Maintaining crop N removal rates of 100–260 kg N year^{-1} for major grain crops (**Table 2**) poses an especially significant challenge given that at the same time we also need to maintain, and often restore, levels of SOM in cropping systems. The importance of SOM for storing carbon, promoting soil biodiversity, and providing a soil structure that improves water use efficiency and drainage is hard to overstate. Moreover, most cultivated soils have already lost much of that portion of SOM that is readily mined for plant-available N (e.g., Reference 5).

Historical Pathways for Supplemental Nitrogen

In traditional subsistence agriculture, rotations were the primary basis for sustained crop productivity. Crops grown on recently cleared land effectively mined SOM as a principal N source. Once SOM stores were depleted to the point of substantially depressed yields, the field was abandoned to pasture or a bush fallow made up of native vegetation that included N$_2$-fixing species capable of restoring N fertility over the course of several decades, when SOM and N stores became sufficient to justify clearing again for cropping (e.g., Reference 6). This shifting cultivation system is sustainable but inefficient in its use of land and thus generally is incapable of producing a surplus that can feed an urban society.

Continuous cropping is much more efficient in terms of land use, but sustaining cropping requires replacement of N losses. Grain producers generally have three options for replacing lost N: (*a*) include N$_2$-fixing crops in the rotation; (*b*) return some portion of the removed N back to the field in the form of manure or human

waste, often supplemented with manure that is ultimately derived from pastures; and/or (*c*) add synthetic fertilizer N. Soybeans (*Glycine max*) represent the most widely grown N_2-fixing grain worldwide; they are commonly grown in rotation with corn and wheat (*Triticum aestivum*). When unfertilized, soybeans can meet a majority of their own N needs (7) but more typically in unfertilized arable soils they fix ~50% of their N (8).

More importantly, only a modest amount of the N_2 that soybeans fix contributes to other crops in the rotation. For example, a soybean crop that yields 4 MT of grain ha^{-1} will remove 250 kg N and leave behind ~7.7 MT of above ground residue containing 70 kg N. This quantity of N can meet only part of the needs of a subsequent corn or wheat crop (**Table 2**). An additional option is to grow legumes as cover crops between the time that a primary crop is harvested and the next is planted (e.g., over winter where moisture is adequate). Winter cover crops are used commonly and successfully in high-value organic agriculture, but their additional cost has precluded their widespread adoption elsewhere.

Manure inputs can provide N where animal agriculture is practiced in concert with grain production. Relatively little ingested N is assimilated by poultry or livestock; most is excreted and can be returned to the field to be recycled into a subsequent crop. This mixed cropping model is one of the most N-efficient farming systems at small scales; in many areas of preindustrial Europe, North America, and China, it was responsible for sustained crop production over centuries. N return is never 100% efficient, however, and thus sustaining this system requires drawing upon manure N that is ultimately derived from nearby pastures as well as cropped lands, potentially degrading their N capital in the process, or from supplementation with N_2-fixing crop rotations or synthetic-N additions. Allen (9) estimates that about half of the yield increase during Britain's agricultural revolution (1300–1800), during which grain yields doubled, was due to the introduction of legumes, such as peas, beans, and clover, into widely adopted forage-grain rotations.

Widespread inclusion of leguminous forage crops and recycled manure into many modern cropping systems is precluded by the increasingly vertical structure of farming, which concentrates animals in confined animal feeding operations (CAFOs) that provide economies of scale with respect to profits but not with respect to nutrient management. Animals in CAFOs are fed diets that are necessarily grown far from the facility, as crop production nearby is not sufficient to meet animal production needs. Manure is bulky and therefore expensive to transport back to where crops were grown, so it becomes a liability rather that a resource, and its overapplication on fields near CAFOs transforms much of its N into pollutants. Effectively, this spatial disconnect removes manure from the N management portfolio of the grain farmer, who must turn to cheaper-to-transport synthetic N in order to replace the grain N shipped to the animal facility. This scenario also holds for the increasing proportion of grain shipped to biopolymer and ethanol refineries: N-rich industrial waste, such as dry distillers grain, is commonly shipped to a CAFO distant from the field of origin, sometimes even overseas.

Synthetic fertilizer is thus the N source of choice for most farmers managing intensive cropping systems: It is easy to transport, readily available, and relatively inexpensive, even at the record costs in 2008 of US\$607 MT^{-1} for urea (10). For example, to apply an additional 10 kg N ha^{-1} cost US\$4.90 in 2000 and US\$13.50 in 2008, but the additional corn yield needed to cover this cost was only 64 kg (2.5 bu) and 69 kg (2.7 bu) at 2001 and at 2007 grain prices, respectively. Overall, the increase in applications of synthetic N fertilizers from ~10 Tg (1 Tg = 1 million tonnes) N/year in the late 1950s to ~100 Tg N/year in 2008 (**Figure 1**) played a central role in the ability of intensive agriculture to increase the rate of food production more rapidly than that of human population growth, as the global population increased from 2.9 billion in 1958 to

6.7 billion in 2008. Neither N use nor increased yields were distributed evenly across the world, and the 10-fold increase in synthetic N applications has been costly in multiple ways. However, the avoidance of human misery resulting from increased food production—an increase that was substantially dependent of additions of N—is an accomplishment that should not be slighted.

THE ENVIRONMENTAL COST OF NITROGEN ADDITIONS

Although the benefits of adding fertilizer N to agricultural systems are straightforward, they are accompanied by substantial costs. These costs arise in part because most of the N added to agricultural systems does not reach its ultimate target—protein in the human diet. For example, of the ~12 Tg of N that are applied to U.S. agricultural systems in fertilizer each year, only ~2 Tg of N are consumed by people, and some of that derives from sources other than fertilizer (e.g., pelagic fish). The 10 Tg of unutilized N is wasted from farmers' and society's perspectives. Moreover, reactive N is highly mobile in the biosphere, and losses from agricultural systems can alter downwind and downstream ecosystems substantially. Finally, the scale of human alteration of the global N cycle is remarkable; total industrial fixation of N amounted to 121 Tg in 2005 (11). Together with N fixed biologically in managed crops or fixed or mobilized during fossil fuel combustion, human activity fixes more N and brings it into circulation in the terrestrial biosphere each year than do all natural processes combined (12, 13).

Hypoxia in the Coastal Ocean

One of the most dramatic illustrations of the environmental cost of excessive reactive N is the massive hypoxic (low dissolved oxygen) zone in the Gulf of Mexico, where runoff from the agricultural heartland of the United States drains through the Mississippi River system to the ocean. This hypoxic zone (popularly called a "dead zone") results from an interaction between fresh water inputs to the Gulf that enhance stratification of the coastal ocean and the supply of nutrients in that fresh water. High-nutrient levels stimulate algal growth, and when algae sink into deeper water and die, their subsequent decomposition by bacteria consumes dissolved oxygen deep in the water column faster than it can be replenished from the surface, leading to the development of hypoxia and the reduction or elimination of deeper-water organisms that require oxygen (14).

Although hypoxic zones can develop without the direct human alteration of nutrient supply (15), paleoindicators of past ocean processes suggest that hypoxic events were sparse or absent in the Gulf of Mexico early in the 1900s when fertilizer use in the Mississippi basin and concentrations of nitrate and phosphate in river water were low (14). Today the size of the hypoxic zone has grown to >20,000 km^2, and there is good evidence that eutrophication—the overenrichment of the Gulf waters that drives the formation of the hypoxic zone—is caused by reactive N rather than P or other nutrients, and good evidence that most of the reactive N in Mississippi River water derives from agricultural systems in the Mississippi basin (16, 17).

Excess N in runoff draining agricultural areas is not confined to the Mississippi River (18, 19), and neither is the development of eutrophic and hypoxic zones in the coastal ocean. Even though some of these areas (e.g., the Baltic Sea) appear to be affected more by P than by N inputs (20), and others (especially rivers draining urban and industrial areas) are influenced by sewage and industrially mobilized N more than by agriculture, agriculturally derived N has a widespread, and rapidly expanding, influence on the dynamics of coastal ocean systems. More than 400 hypoxic areas have been identified, and their frequency has approximately doubled each decade since the 1960s (21).

Fates of Reactive Nitrogen

Hypoxic zones, algal blooms, and other effects on the coastal ocean are driven by transfers of

soluble reactive N from terrestrial ecosystems (often agricultural fields) to streams, rivers, and ultimately the ocean. Losses in solution to rivers and oceans represent an important, but far from the only, fate of excess reactive N lost from agricultural systems. Some soluble N is transformed into organic or volatile forms along the flow path from fields to ocean; some of these forms are benign, and some cause further environmental problems. Some portion of soluble N leaches deep into groundwater, ultimately affecting human health (e.g., Reference 22). In the United States, about 20% of groundwater supplies in agricultural regions exceed the U.S. Environmental Protection Agency's established health limits of 10 mg NO_3^--N/L (nitrate-N per liter), and in other regions of the world, elevated nitrate contamination also tends to be associated with N fertilizer use (23–25).

In addition to solution losses, reactive N can also be lost to the atmosphere as different kinds of N-containing gases (**Table 1**). Gaseous N forms range from N_2, which is environmentally benign, to the radiatively active gas N_2O, which contributes to climate forcing, to nitric oxide (NO) and nitrogen dioxide (NO_2) (collectively known as NO_x), which contribute to oxidant air pollution and acid rain, to NH_3, which can transfer large quantities of reactive N to downwind ecosystems.

Nitrous Oxide and Climate Change

Concentrations of nitrous oxide (N_2O) are increasing in the troposphere; they were stable near 270 ppbv (parts per billion by volume) for thousands of years until the past two centuries but have since increased continuously to ~320 ppbv (**Figure 3**). N_2O is not reactive in the troposphere, but it absorbs outgoing infrared radiation from Earth and so acts as a powerful greenhouse gas, one that is approximately 300 times more effective than carbon dioxide (CO_2) on a molecule-by-molecule basis. N_2O contributes ~6% to the total greenhouse gas forcing that drives climate change (26). Global budgets for N_2O (e.g., Reference 27) conclude that increasing concentrations

reflect an anthropogenic source of 5–6 Tg of N_2O-N. About 80% of this source is associated with agriculture, largely (50%) with fertilized soils (28). A more recent analysis by Crutzen et al. (29) concludes that the recent increase in atmospheric N_2O concentrations could be explained if only ~4% to 5% of newly created reactive N ultimately is emitted to the atmosphere as N_2O; they demonstrated that emission of a similar fraction of biologically fixed N in the preindustrial era could explain the preindustrial terrestrial source of N_2O. Because most anthropogenic reactive N is produced in or for agricultural systems (13), most of the increase in N_2O is due (ultimately) to agricultural uses of N.

At ~6% of global greenhouse gas forcing, N_2O is not one of the most important drivers of climate change. However, it plays a substantial role in the agricultural contribution to climate change, and its emissions can offset efforts to use agricultural systems to mitigate climate change by sequestering CO_2 or providing alternative energy sources (29, 30). Moreover, when converted to carbon (C) equivalents using 100-year global warming potentials, the anthropogenic N_2O flux is equivalent to 1.0 Pg C year^{-1} (28, 31), which compares to the contemporary net atmospheric CO_2 increase of 4.1 Pg C year^{-1} (32).

Reactive Nitrogen Gases

In addition to N_2O, agricultural systems emit reactive N gases, particularly NH_3 and oxides of N. NO is the most abundant oxide emitted from agricultural and other soils, and it oxidizes quickly in the atmosphere to form NO_2 and other compounds. Fluxes and atmospheric budgets of reactive oxidized N generally are reported in terms of NO_x, the sum of all of such forms.

The emissions of both NO_x and NH_3 have been increased substantially by human activity, including agriculture. As of the mid-1990s, ~80% of terrestrial NO_x emissions were anthropogenic (13); about three-quarters were from energy systems, and the other one-quarter
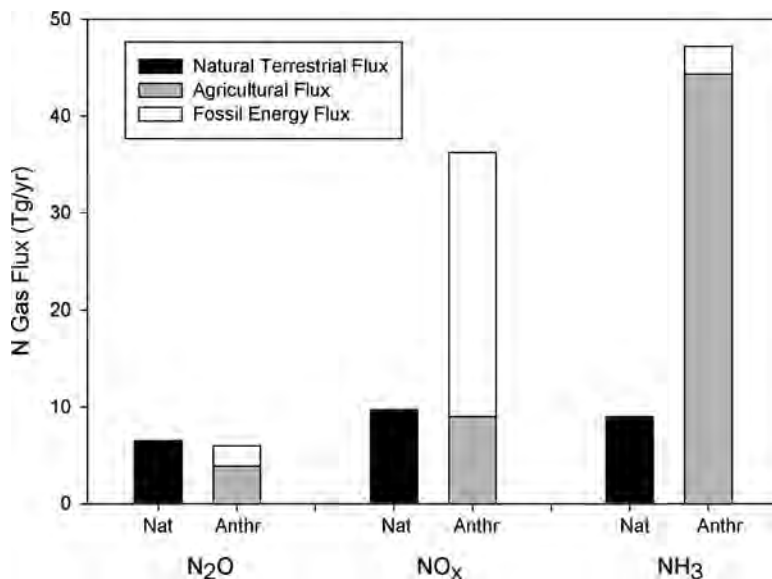
Figure 4

Natural (Nat) and anthropogenic (Anthr) sources of nitrogen (N)-containing trace gases from terrestrial ecosystems. Agriculture (broadly defined to include land clearing, agricultural burning, and animal operations) is responsible for most ammonia emitted to the atmosphere. Agriculture also produces about as much NO_x as all natural sources combined (though much less than fossil fuel sources), and it makes a substantial contribution to increasing concentrations of the greenhouse gas N_2O in the atmosphere. All fluxes as of the early 1990s, from Reference 13. Abbreviation: NH_3, ammonia.

was from agriculture, broadly defined to include agricultural burning and land clearing as well as fluxes from agricultural soils (**Figure 4**). NO_x plays a substantial role in tropospheric photochemistry; when NO_x is elevated, the oxidation of atmospheric hydrocarbons and carbon monoxide lead to the production of ozone (O_3), but when NO_x concentrations are low, O_3 is consumed (33, 34). Tropospheric O_3 is both a greenhouse gas and an oxidant that harms human health (22) and plant growth at concentrations that are reached frequently in high-NO_x regions.

Although urban and industrial sources of NO_x are more important globally than agricultural sources, several tropical regions experience high NO_x and O_3 concentrations, at least in part as a consequence of agriculturally based biomass burning (35). Elsewhere, the widespread intermingling of urban, industrial, and agricultural land uses in "metro-agroplexes" (36), within which the majority of NO_x

emissions take place, makes it challenging to separate our management of sources of oxidant air pollution into industrial and agricultural components. Eventually, NO_x emitted from energy or agricultural systems is deposited on downwind ecosystems in gaseous [NO_x and nitric acid (HNO_3) vapor], particulate, or dissolved (HNO_3 and oxidized organic N) forms. Most is deposited from hundreds to ~2000 km downwind from where it is emitted (37).

Unlike NO_x, a very large fraction of terrestrial NH_3 emissions are from agricultural sources (**Figure 4**). In the mid-1990s, close to 95% of anthropogenic NH_3 emissions were from agricultural systems—about half derived from animal wastes and a quarter from fertilizer applications (**Figure 4**) (13). NH_3 plays a relatively benign role in atmospheric chemistry, with most emissions deposited within 1000 km downwind as NH_3, or as ammonium (NH_4^+) in rainwater or aerosols. In all, Galloway et al. (13)

calculate that 53 Tg of anthropogenic reactive N were emitted from agricultural systems to the atmosphere in 1993, more than 80% as NH_3, most of which was deposited within and downwind of regions of intensive agriculture. Another 30 Tg of anthropogenic reactive N were emitted from energy systems, more than 90% as NO_x; the areas where NO_x and its products were deposited overlap strongly with areas of NH_3 deposition (37).

Consequences of Nitrogen Deposition

Transfers of reactive N from agriculture to downwind ecosystems can alter the functioning and biological diversity of the recipient systems—just as solution losses of reactive N influence the functioning and diversity of downstream and coastal ecosystems (14). Indeed, a recent analysis of marine systems concluded that atmospheric deposition of anthropogenic reactive N to open-ocean systems amounted to 67 Tg $year^{-1}$ of N in 2000, of which 54 Tg were anthropogenic (38). This input represents a substantial, and unevenly distributed, perturbation to the N economy of the open ocean.

On land, the most significant consequences of N deposition stem from the limited supply of fixed N in many ecosystems (39, 40), which makes them vulnerable to alteration by additional N, and by the acidifying effect of most forms of atmospheric deposition. Deposition of NH_4^+ is acidifying to soils in that NH_4^+ uptake by plants is balanced stoichiometrically by rhizosphere H^+ formation (41, 42), and bacterial nitrification of NH_4^+ to nitrate (NO_3^-) produces two moles of H^+ for every mole of NO_3^- (43). Where NH_4^+ is transformed to NO_3^- within ecosystems, or where HNO_3 is deposited, mobile NO_3^- can transport acidity to downstream ecosystems.

N deposition to N-limited ecosystems increases productivity, which, on the positive side, could sequester some of the ongoing increase in atmospheric CO_2 as plant biomass C. However, enrichment with N often drives substantial changes in species composition and overall declines in plant species diversity (41). For example, northwestern Europe receives some of the highest levels of N deposition, mostly from agriculture, leading to widespread changes in heathlands and other ecosystems of infertile soils (44). Although most plant species in N-poor ecosystems would respond positively to added N in isolation, typically a few species (often grasses) respond most strongly, monopolize space and other resources, and drive an absolute decline in species diversity. Typically, rare species and those belonging to certain defined functional groups (such as N fixers) are most likely to be eliminated (45, 46). Parallel changes in composition and diversity are observed as a consequence of experimental N fertilization under controlled conditions (47).

The compositional and functional changes induced by added N are not confined to plants; both observations of areas altered by N deposition (48) and controlled experimental studies (49, 50) document changes in soil fungal communities following additions of reactive N. Plant and soil communities appear to be most responsive to added N where it is in particularly short supply, as in many boreal forest ecosystems (51). In addition, the influence of N deposition is multiplied where it interacts synergistically with other components of anthropogenic change. For example, enhanced N deposition in Southern California facilitates invasion of Mojave Desert ecosystems by introduced grasses, notably red brome (*Bromus rubens*) (52); that invasion, in turn, facilitates the spread of fire and a complete transformation of the desert shrubland into a desert grassland (53).

Where N is not in short supply, either naturally or as a consequence of past anthropogenic enrichment (54), the major consequences of enhanced N deposition include acidification, cation depletion, and in extreme cases dieback of dominant species (54, 55). Enriched systems also lose more reactive N to the atmosphere and downwind ecosystems as well as downstream—thereby expanding the source area for enhanced transport of N through the biosphere (56).

BASIC NITROGEN-CYCLING PROCESSES

The fundamental challenge in agricultural N management is to enhance agricultural productivity to reduce hunger, feed a growing population, and support changing demands for food—while simultaneously reducing the transfer of reactive N to nontarget ecosystems. The challenge is easy to state and is one for which the economic incentive to use fertilizer N sparingly should align directly with the environmental benefits of avoided N additions to downwind and downstream ecosystems. However, although improvements that jointly address farm profitability, yields, and environmental benefits have been identified and implemented in particular cases (57, 58), the overall challenge has proved a difficult one. In most intensive agricultural systems, on-farm trials consistently show that less than half of applied N is recovered in crops (4). Much of the remainder commonly is lost to the environment.

Improving the efficiency with which N is used, and managing fates of N so that unavoidable losses move by pathways that cause less environmental harm, requires a deep understanding of the cycle of N in agricultural ecosystems—particularly of the processes that mobilize or retain reactive N. Focused, long-term experiments that evaluate both the overall consequences of management practices and the processes within the N cycle that control those consequences are required (59).

The major processes and transformations in the N cycle of agroecosystems are outlined in **Figure 2**, a complex figure, as befits this most complex of element cycles. Moreover, the figure vastly understates the true complexity of the cycle in terms of the diversity of transformations that are known, the range of mostly microbial populations that carry them out, and the variety of fates of N deposited to or transformed within ecosystems. Nevertheless, it provides a useful summary of the processes controlling sources, transformations, and fates of N as these relate to agricultural ecosystems.

Inputs of Nitrogen

New N is added to cropping systems by N fertilizer (arrow A in **Figure 2**), organic amendments (mulch and manure) (B), biological N_2 fixation (C), and atmospheric deposition of combined N (D and E). A and B are management-controlled inputs; they differ in that most synthetic fertilizer (mainly inorganic N and urea) is immediately or rapidly available for uptake by plants and microbes, whereas organic amendments must be processed by microorganisms before most of its N is available for uptake.

Biological N_2 fixation is carried out by microorganisms in both agroecosystems and little-managed systems. Some of these microbes are free living and obtain their energy from organic matter or, in the case of bluegreen algae, from photosynthesis. More N is fixed by plant root symbionts that obtain energy from photosynthate provided by the plant. The legume-*Rhizobium* symbiosis, for example, is especially important in agriculture; historically, it was used to replace much of the protein N removed in crop harvests. Rhizobia bacteria associated with soybean, alfalfa (*Medicago sativa*), and other leguminous crops can fix large quantities of N_2 (60), and in other cropping systems [rice, sugar cane (*Saccharum officinarum*)] meaningful quantities of N can be fixed by other microbes (61, 62), but most crops fix little or no N directly. However, leguminous cover crops may add substantial quantities of N to any agricultural system (63), and heterotrophic N_2 fixation during decomposition is a widespread if generally minor source of fixed N (64). Atmospheric deposition of fixed N is outside of managers' control, though it must be taken into account in the analysis of nutrient supply to crops in or downwind of intensive agricultural and industrial regions (65).

Nitrogen Transformations

Our understanding of the diversity of N transformations in ecosystems has expanded enormously in recent years, largely as a consequence

of the application of molecular approaches to microbial communities in soils, solutions, and sediments. Some of the key discoveries include the extraordinary diversity of bacteria, fungi, and archaea in soils (66) and the potential significance of that diversity to rates of soil N transformations (67); the broad role of archaea as well as true bacteria in N fixation, nitrification, and denitrification (68–70); and the identification of previously unknown pathways of N transformation (e.g., anammox) (71, 72). Understanding these and other developments in our knowledge of the N cycle ultimately may contribute to nutrient management in agricultural systems; for now, the more aggregated analysis of N transformations in **Figure 2** remains useful.

We suggest that the key N transformations that need to be considered in agricultural ecosystems are (*a*) mineralization (arrows F and G), which takes N from insoluble organic forms to soluble and biologically available forms; (*b*) nitrification (H), which takes N from reduced and less mobile NH_4^+ to oxidized and more mobile NO_3^-; (*c*) uptake by crops and other plants (I); and (*d*) immobilization by microbes (J), which retains otherwise-mobile N against loss to the environment.

Conventionally, N mineralization is defined as the transformation from organic to inorganic (initially NH_4^+) forms; however, recent research makes clear that the rate-limiting step in mineralization typically is the depolymerization of organic N, which is the breakdown of complex, insoluble, N-containing organic compounds by extracellular enzymes produced by microorganisms, leading to the release of soluble and biologically available organic compounds including amino acids (73). Amino acids can be taken up by plants as well as by microorganisms (74), but in the relatively high-N conditions typical of most agroecosystems, most amino acid N is further transformed to inorganic forms before it is utilized by plants (73).

Nitrification oxidizes NH_4^+ to nitrite (NO_2^-) and then to NO_3^-; typically, NH_4^+ oxidation is the rate-limiting step in the overall process, and little NO_2^- accumulates in soils. The process is important because, as an anion, NO_3^- is more easily transported to roots and to groundwater than is the cation NH_4^+, as most agricultural soils have little anion exchange capacity. Equally important, NO_3^- can serve as a terminal electron acceptor in place of oxygen during microbial respiration, whereby NO_3^- is reduced to molecular nitrogen (N_2) and to N-containing trace gases, as discussed below. Also importantly, in most annual crops, N is taken up mostly as NO_3^- (I in **Figure 2**) (75), and without nitrification, little nitrate would be available.

Finally, soil microorganisms often satisfy part of their demand for N via immobilization—the uptake of available N from soil (arrow J in **Figure 2**). On the one hand, demand by microbes may compete with that by plants, reducing the supply of N to crops. On the other hand, microbial immobilization retains otherwise mobile N within soils, where later N mineralization can make it available to subsequent crops.

Nitrogen Outputs

Major pathways of N loss from cropped ecosystems include harvest of agricultural products (arrow K in **Figure 2**), leaching to surface and groundwater (L), denitrification to N_2 (M), volatilization of NH_3 (N), and fluxes of N_2O and NO_x to the atmosphere (O and P). Although not specific to the N cycle, erosion by wind and water (not shown) also removes multiple forms of N from agricultural ecosystems (76). In addition, fire (also not shown) can remove N as NO_x, N_2O, and N_2 where crop residues or fallow vegetation are burned prior to cultivation.

Leaching of NO_3^- typically is one of the three largest losses to the environment in most upland cropping systems; while some dissolved organic N and NH_4^+ are lost to surface and groundwater, the mobility of NO_3^- ensures that it is normally the dominant form of dissolved N in water moving through agricultural soil profiles. NH_3 volatilization is a second major pathway by which N can be lost from agroecosystems; NH_3 is lost from animal production

systems, soils, and plants. NH_3 is in pH-dependent equilibrium with NH_4^+ in soils and solutions; most losses by volatilization from soils occur shortly after fertilization when both NH_4^+ abundance and pH can be high. Losses can be especially high when liquid manure, urea, or anhydrous NH_3 are applied under suboptimal, dry soil conditions. Losses of NH_3 from plants directly to the atmosphere can also occur, especially during senescence (77).

Denitrification to N_2 removes biologically available N from agricultural ecosystems, but because it returns N to its massive, little-reactive reservoir in the atmosphere, its environmental consequences are benign. Ideally, excess N applied to agricultural systems would go through this pathway, and indeed, denitrification (together with ocean sediment burial) closed the global N cycle in preindustrial times and thereby kept reactive N from accumulating in the biosphere. However, a fraction (typically a small one) of the N moving through both the nitrification and denitrification pathways is emitted to the atmosphere as NO_x and N_2O—with relatively more NO_x from nitrification (P in **Figure 2**, coming from arrow H) and more N_2O from denitrification (O in **Figure 2**, coming from the nitrate pool). The quantities involved are not large relative to overall N losses from agricultural systems, but they make meaningful contributions to global NO_x emissions and major contributions to N_2O fluxes (**Figure 4**) as discussed above.

Denitrification is technically difficult to quantify in agricultural systems; more often than not it is estimated by difference upon quantifying all other fluxes—as was done 50 years ago (3)—or by modeling. We thus have less direct knowledge about the magnitude of this major flux than about any other (30). Nevertheless, except in flooded soils, denitrification losses of N appear smaller than losses via nitrate leaching, but denitrification in riparian zones, sediments, and wetlands downstream from agricultural fields can redirect much of the N that leaches from agroecosystems to the atmosphere as N_2.

NITROGEN USE EFFICIENCY: CONCEPTS AND CHALLENGES

How can the benefits of N management be sustained and expanded while minimizing the multiple threats to the environment caused by losses of reactive N to downstream and downwind ecosystems? At its core, the challenge is a familiar one to agronomists—how can the efficiency of N additions to agriculture be increased though better management of inputs, crops, N-cycling processes in fields, and agricultural landscapes while maintaining high crop yields. Although N use efficiency (NUE) appears to be a simple concept, in practice, there are multiple ways of assessing NUE and multiple pitfalls to their application. Not the least of these challenges is the timescale over which measurements are needed.

The most widely applied measures of NUE evaluate the yield per unit of added N; for example, agronomic NUE (AE_N) is defined as:

$$AE_N = (Y_f - Y_0)/N_{rate},$$

where Y_f is yield in fertilized areas, Y_0 is yield in unfertilized subplots, and N_{rate} is the rate of N addition (78). Alternative measures evaluate NUE in terms of N rather than yield, considering the fraction of applied N that is taken up by crops or removed in harvested products. We make use of these N-based measures here.

There are two main ways of calculating NUE in terms of N balances. In both cases, NUE is higher where a greater proportion of added fertilizer N is captured by the crop.

1. Labeling studies make use of N additions that are labeled with the stable isotope ^{15}N; the recovery of applied ^{15}N in crops or in harvested products is contrasted to the amount of ^{15}N label added (^{15}N label recovered/^{15}N label added).
2. Mass balance studies compare the quantity of N removed from agricultural fields in harvested products with the amount of N added by managers (total N harvested/total N applied).

The first of these approaches is especially useful for understanding N-cycling processes in agricultural systems because it allows us to differentiate N added by managers from N already present in the system (or added by uncontrolled pathways). Typically, however, it understates longer-term NUE because some of the added label is immobilized in the year of application and then mineralized (and harvested) in subsequent years.

The second approach also presents challenges in interpretation: For NUE to be represented fairly, the N added by managers must represent the main source of N entering agricultural systems, and the amount of organic N in soil has to be near steady state. Where SOM is declining significantly, NUE will be overstated. However, it has the advantages of being based on widely available information, of being readily integrated over multiple years, and of avoiding the time lags associated with direct measurements (79). The last is particularly important; annual measurements of NUE by any metric (including labeling or yield-based ones) are highly dependent on growing conditions in the year of measurement and on both management practices and growing conditions in previous years.

The amount of N added to agricultural fields often is substantially out of balance with the amount removed in harvested products, and the difference can occur in either direction. For example, intensively cultivated corn-based systems in western Kenya receive only 11 kg N ha^{-1} of fertilizer plus manure; 59 kg N ha^{-1} is removed in grain and secondary harvests, leaving a net imbalance of -48 kg N ha^{-1} $year^{-1}$ (80). These systems function only by exploiting unsustainably what were once substantial pools of organic N in soil, and the fertility of their soils is degraded in the process. By contrast, in the wheat/corn double-cropping system in the North China Plains, annual fertilizer additions are 588 kg N ha^{-1}, plus 61 kg N ha^{-1} of added manure. Only 361 kg N are removed in yields, leaving a net positive N balance of 288 kg N ha^{-1}, most of which is lost via nitrate leaching and NH_3 volatilization (58). Many U.S. systems

have similar if less extreme patterns of positive balances as in China, and so smaller but still substantial N losses.

Typically agricultural system N balances in the developed or rapidly developing worlds are positive, with N additions in excess of harvest N removals—a pattern that implies substantial losses of N to the environment—with all of the attendant costs. How can N delivery to crops—and crop yield—be sustained or enhanced while minimizing losses of N to the environment, especially losses by more damaging pathways? Conceptually, the simple answer is that synchronizing the supply of available N (whether through additions of available N in fertilizer, N_2 fixation, or via mineralization of organic amendments or SOM) to biological demands for that N should maximize the retention and use of N. This synchronization of supply and demand is more typical of intact, unmanaged systems; there, efficiency is achieved by the long seasonal duration of plant N uptake and the diversity of plant life history strategies used to capture N. Moreover, plant species in diverse natural systems vary with respect to their ability to remove N from the soil solution (75) and to explore different parts of the soil profile (81).

In contrast, N is taken up at significant rates for only 8–12 weeks following crop canopy closure in most annual cropping systems (82). Although these rates can be extraordinarily high, approaching 2 kg N ha^{-1} day^{-1} for corn, they drop to nil at crop maturity and do not recover until the following growing system. Consequently, N that is available, but not taken up by the crop during its growth phase, plus N that becomes available to the system during other times of the year—for example, that derived from the mineralization of N in soil and organic amendments in the fall, winter, and much of the spring—can be lost to the environment.

Much of the low N efficiency of annual cropping systems thus is derived from their short growing season and simplified design, and the addition of fertilizer N exacerbates their leakiness. Such systems lose N owing to (*a*) the inability of annual crops to remove N from soil efficiently; (*b*) the tendency of most producers

with access to fertilizer to add more N than is required for optimum yield, in order to minimize perceived economic risks; and (*c*) the inherent difficulty of supplying external N economically to plants. Even where crops take up as much as 2 kg N ha^{-1} day^{-1}, typical increments of fertilizer applications are 50 times this (or more), making most of each application vulnerable to loss well before it can be taken up by crops.

IMPROVED NITROGEN MANAGEMENT: TECHNOLOGY POTENTIALS

Many approaches for improving the NUE of high-productivity agricultural systems have been identified; one recent paper (79) provided a nonexclusive list of 44 such methods. These approaches fall into four main options for annual cropping systems: (*a*) adjust the crop rotation to add complexity that improves the plant community's ability to take up more available N; (*b*) provide farmers with decision support tools that allow them to better predict crop N requirements and avoid overfertilization and to better schedule irrigation to avoid driving N below the root zone; (*c*) better manage the timing, placement, and formulation of fertilizer N in cropping systems to ensure N is available where and when plant demand for N is greatest; and (*d*) manage watersheds to mitigate or redirect N losses downstream from fields. A fifth set of options applies primarily to intensive animal agriculture. All of these strategies are likely to require incentives to promote their adoption.

Rotational Complexity to Conserve Nitrogen

The simplest if often most expensive means for capturing more of the N added to annual cropping systems is to include cover crops in a rotation. In temperate regions with adequate moisture, winter annuals such as ryegrass (*Lolium multiflorum*) can be planted in the fall following harvest or even overseeded into the principal crop during the growing season. Fall growth captures some proportion of the residual N that

remains from the summer crop (83, 84), and after winter dormancy, the cover crop is ready to grow rapidly with the onset of springtime temperatures that also stimulate N mineralization from crop residue and SOM. Prior to planting the next principal summer crop, the cover crop is killed, and its subsequent decomposition releases a portion of the N captured in plant biomass since the preceding fall.

Cover crops do not provide complete protection against N loss; prior to establishment in the fall, N remains available for hydrologic and gaseous loss, as it does between the time that the cover crop is killed in the spring and the principal crop closes canopy. Nevertheless, ensuring that at least some plant uptake occurs during times of the year when decomposition and N mineralization are active is a meaningful way to conserve N within the cropping system and thereby improve system-level NUE.

Leguminous cover crops can be even more advantageous because of the additional N that they can accrue from N$_2$ fixation. Because N$_2$ fixation will be low when adequate soil N is available, winter legumes can provide the same degree of soil inorganic N scavenging as their nonleguminous counterparts. They have the additional advantage of producing biomass with a low C:N ratio so that decomposition of the spring-killed residue is rapid, which makes more N available earlier for the growth of the principal crop in early summer (85, 86). Moreover, cost savings from avoided N fertilizer use can be put toward cover crop seed and planting expenses.

The associated benefits of cover cropping include building SOM and, in regions with wet springs, helping to dry early spring soils, which can expedite spring planting. Disadvantages include the added expense of planting and killing the cover crop as well as the risk that spring growth might deplete stored soil moisture in the absence of adequate springtime precipitation. For this reason, cover crops are less suitable in drier regions, where no-till management may be the more important management intervention for preventing off-season N loss because it protects soil against losses via erosion.

Intercropping, in which two crops are grown simultaneously, is another means for increasing NUE through rotational complexity. Intercropping is rare in mechanized row-crop agriculture owing largely to engineering challenges associated with the need to harvest crops at different times without damage to the remaining crop. The alternative is to plant one of the crops as a cover crop only, but this creates competition for limited resources such as water and N, potentially obviating any advantage to increased NUE. Nevertheless, in specialized cases, intercropping can be advantageous: In tropical agroforestry (87), for example, trees can pull N from deeper in the profile to be recycled when leaf litter is dropped onto surface soils, and in mechanized fields, strip crops (88, 89) can be planted downslope and thereby capture N leached from upslope positions that would otherwise be lost downstream.

Additionally, some annual crops have lower N needs and are more efficient at scavenging N from soil than are others, so including a mixture of crops with different N requirements and scavenging abilities in a multiyear rotation will increase the rotation's NUE. Winter wheat, for example, typically requires less N and captures more of that applied than does corn. This is partly because less N is removed in wheat yield (**Table 2**); partly because fall-planted wheat can scavenge residual N that remains in the soil from the previous crop; and partly because when fertilizer N is applied the following spring the wheat is well established and actively growing, leaving soil N less available for fates other than crop uptake. A corn-wheat rotation thus can have a higher NUE than a continuous corn rotation.

Finally, the development of crop varieties with higher efficiencies of N uptake could help capture more of the N added to annual cropping systems. Preliminary evidence that hybrid isolines of corn containing the Bt rootworm trait have a higher NUE than standard varieties (90), together with developments in understanding the genes controlling NUE and its component traits (91), suggests that varieties with enhanced uptake efficiencies might make meaningful contributions to system-level NUE in the coming decades.

Better Prediction of Crop Nitrogen and Water Needs

Since the 1970s, most N fertilizer recommendations in the United States have been based on yield goals, an approach that stems from the call for using mass balance to assess crop N needs (92). The yield-goal approach provides an N fertilizer recommendation for a particular field on the basis of an expected maximum yield for the field's soil type multiplied by an N yield factor. For corn on a moderately high-yield soil in the U.S. Midwest, for example, a yield goal of 10 MT/ha (160 bu/acre) would be multiplied by 21 kg N/MT (1.2 lb N/bu) to provide a recommendation of 210 kg N ha^{-1} (192 lb N/acre).

Soil N tests prior to fertilization can often substantially improve yield-goal N recommendations, particularly when legumes or other nonfertilizer N inputs are in the rotation. In a 10-year study of >100 N response field experiments in the upper Midwest, for example, Andraski & Bundy (93) found that a pre-sidedress soil nitrate test (PSNT) (94) could have reduced recommended rates of fertilizer N by 92–102 kg N ha^{-1} year^{-1} for systems where manure had been applied or legumes grown within the previous three years, without any loss of expected yield. However, prefertilization soil tests have not been effective predictors of future N needs in many areas where they have been tested.

Without a widespread reliable soil test to predict soil N mineralization, variation in soil N supply (mineralizable soil N) provides a significant challenge to the yield-based approach, as does uncertainty about how yield goals should be determined (95). An alternative approach now being adopted in much of the U.S. Corn Belt is based on the site-specific N rate at which the value from increased grain yield just matches the cost of added N. Any additional N cannot be economically justified in the absence of higher grain prices or cheaper fertilizer. This methodology involves constructing N response

curves for various cropping systems on different soils. By definition, the economically optimum N rate will be lower than the fertilizer rate at which yields are maximized (**Figure 5**). How much lower is determined by the ratio of N price to grain price; as N becomes more expensive or grain price declines (increasing the ratio), producers will reap the same profit with less fertilizer N. Calculating separate economic optimums for different rotations, e.g., one for corn following corn and another for corn following soybean, removes the need to estimate a residual N credit for the preceding crop.

In irrigated crops, which include 8% to 10% of cropland in the United States and Europe (96), an additional management intervention for N conservation is better irrigation scheduling. Applying sufficient water to satisfy crop needs without leaching NO_3^- below the rooting zone can be as important for NUE as better N rate recommendations. Gehl et al. (97), for example, found that for corn in the U.S. Midwest, irrigation at rates only 25% higher than optimal exacerbated the amount of NO_3^- leached to groundwater more than an order of magnitude. Because most irrigated field crops are high value, they also tend to be amply fertilized, making proper irrigation especially important for N retention under irrigated conditions.

Fertilizer Timing, Placement, and Formulation

Mismatched timing of N availability with crop need is probably the single greatest contributor to excess N loss in annual cropping systems. Ideally, N ought to be applied in multiple small doses when plant demand is greatest. However, except where N can be applied in irrigation water, the timing and minimum quantity of fertilizer application is dictated by weather and the availability of equipment and labor. Commonly, best practice calls for two applications to field crops, such as corn, with a starter rate ($\sim$30 kg N ha^{-1}, for example) applied at planting and a side-dress rate (the remaining N to be applied) several weeks later, once the crop has

germinated and entered a rapid growth phase. Where fertilizers are not side-dressed, a single fertilizer application is made in the spring, or even in the previous fall (where producers need to minimize the number of field operations in the spring). Much of the N applied to Iowa corn, for example, is applied as anhydrous NH_3 in the fall, leaving 8–9 months for fertilizer N to be lost to the environment prior to crop uptake. Fall- and winter-applied manure is subject to the same fate.

Fertilizer placement and method of application also are important. The spatial arrangement of plants, additions of N, and other resources within a field may be mismatched, and a poor match will promote N loss. From a management perspective, there are two important scales of heterogeneity—row-interrow variability and heterogeneity across whole fields. Row-interrow differences in N availability and turnover are well known (e.g., References 98 and 99), and a number of management strategies based on these differences have been used to increase water and nutrient use efficiency in row crops. Drip irrigation, for example, is used in high-value vegetable, citrus, and sugar cane crops to place water and N close to plant roots, and fertilizer banding—placing fertilizer in a concentrated band within or very close to crop rows rather than between them—is common in field crops and can substantially increase NUE (100). Ridge tillage and the formation of seed beds prior to planting are additional techniques for concentrating available nutrients and organic matter in crop root zones.

N availability is highly variable in natural communities, with patches of more decomposable organic matter leading to variation in N mineralization at scales that can affect individual plants (e.g., Reference 101). Long-term cultivation appears to attenuate and enlarge preexisting patches rather than remove them (102), underscoring the persistence of ecological legacies related to past land use. The net result of these legacies plus heterogeneity related to underlying geomorphological features is field-scale variability that is a major feature of all cropping systems. Consequent patterns of crop
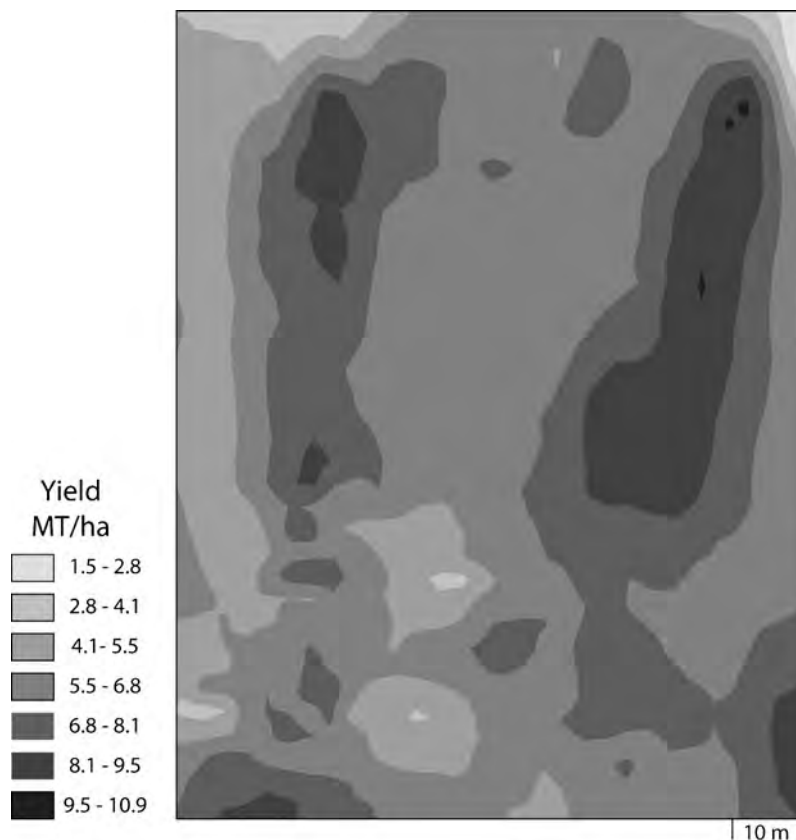
Yield
MT/ha

	1.5 - 2.8
	2.8 - 4.1
	4.1 - 5.5
	5.5 - 6.8
	6.8 - 8.1
	8.1 - 9.5
	9.5 - 10.9

10 m

Figure 6

Variation in yield across a 1-ha portion of a southwest Michigan corn field following a late-season drought.

productivity (e.g., **Figure 6**) have fundamental implications for field-scale NUE.

Consider, for example, crop yields that vary by a factor of 2 across a 50-ha field: for any amount of fertilizer applied evenly to the entire field, high-yield areas will have a higher NUE than low-yield areas. This means that the common management decision to maximize yield will substantially overfertilize those portions of the field with low-yield potential. If one-third of the field is responsive to 150 kg N ha^{-1}, another third to 120, and the remaining third to only 80, then fertilizing the entire field at 150 kg N ha^{-1} would lead to two-thirds of the field's receiving at least 30–70 kg ha^{-1} excess N, all of which would likely be lost to the environment.

Precision agriculture technology provides the potential to avoid much of this mismatch, in

that applying fertilizer to a field at variable rates selected to coincide with crop production potentials as they vary in space could substantially increase fertilizer use efficiency. For example, Scharf et al. (103) found optimal rates of N fertilizer for corn varied substantially within seven of the eight U.S. Midwest corn fields they examined, with a range 0–280 kg N ha^{-1} for five of the fields; they concluded that had each field been fertilized at its median optimal rate, 25% of each field would have been overfertilized by 34 kg N ha^{-1} and 10% overfertilized by 65 kg N. Had each been fertilized at its maximum recommended rate (up to 280 kg N ha^{-1}), overfertilization would have been even higher. For a Minnesota field, Mamo et al. (104) estimated that variable N fertilizer application at rates up to 145 kg N ha^{-1} would have avoided the use

of 69–95 kg fertilizer N ha^{-1} as compared to fertilizing at the recommended uniform rate of 145 kg N ha^{-1}. On-the-go fertilization (105), whereby variable-rate N applications to a growing crop are made on the basis of real-time spectral reflectance of the crop canopy, offers a promising new technology for matching N inputs even more precisely to need.

Advanced fertilizer formulations and microbial inhibitors provide another means for better delivery of N to plants. Synthetic fertilizer is sold in a variety of forms and formulations; worldwide, the most common form is granular urea, which accounts for more than 50% of world fertilizer consumption owing to its low production and transportation costs. Urea is rapidly converted to biologically available N once applied to soil by the following pathway:

$$(NH_2)_2\,CO + H_2O \xrightarrow{\text{urease}} 2NH_3 + CO_2.$$
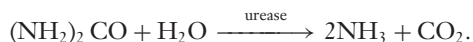
The initial reaction of urea with soil enzymes produces NH_3 gas, which is readily lost to the atmosphere. However, in most soils, NH_3 from urea that is applied appropriately (rapidly incorporated or injected into nonalkaline soils) will quickly hydrolyze to NH_4^+, which is soluble and therefore retained in the soil solution. In turn, though, in arable soils, NH_4^+ will quickly nitrify to NO_3^-, which is also soluble but extremely mobile and subject to loss prior to plant uptake, as described above.

Other common fertilizers are equally subject to environmental loss: Soluble ammonium from anhydrous NH_3, ammonium sulfate, ammonium nitrate, and solutions of urea ammonium nitrate will also quickly nitrify to nitrate as soon as soil conditions are favorable for microbial activity. Two strategies can delay nitrification and thereby keep N in forms less subject to environmental loss: fertilizer formulations that delay the dissolution of N in soil, deferring the exposure of ammonium to nitrifiers, and soil amendments that inhibit nitrifiers biochemically.

Slow-release fertilizers commonly are pelletized formulations coated with a substance or membrane that slows solubility. Sulfur-coated urea is the oldest of these technologies, produced by coating urea pellets with a layer of molten sulfur that is additionally coated with a sealant such as polyethylene oil or microcrystalline wax. Polymer coatings provide an improved but more expensive barrier; the polymers create a semipermeable membrane and can be applied to granules of a variety of substrates, including urea and ammonium nitrate. Coating thickness and additional sealants provide specific release rates, which are typically temperature and moisture dependent.

Nitrifying bacteria can be inhibited by natural and manufactured compounds. A wide variety of plant extracts inhibit culturable nitrifiers, and a few of these, such as neem oil extracted from the Indian neem tree (*Azadirachta indica*), have been shown to slow nitrification in situ. Manufactured inhibitors such as nitrapyrin [2-chloro-6-(trichloromethyl) pyridine]and dicyandiamide [$H_2NC(=NH)NHCN$] can also slow nitrification when added to fertilized soil, though in practice with mixed success (106). Agronomic evidence that any nitrification inhibitor consistently increases NUE is lacking; the general problem seems to be a combination of delivery of the inhibitor to the microsites where nitrification occurs, persistence of the effective ingredient in soil, and an often narrow range of environmental conditions (in particular moisture, temperature, pH, and texture) over which the inhibitor is effective.

Watershed-Level Strategies for Mitigating Nitrogen Excess

In addition to increasing the efficiency with which N is used in agricultural fields, management strategies can seek to retain N within agricultural landscapes and to redirect losses that do occur into less harmful pathways. Agriculture contributes N to surface waters and groundwater via surface and subsurface water flow, and encouraging stewardship at the scale of watersheds can reduce losses of reactive N substantially (107), particularly as follows:

1. Riparian and other downslope conservation plantings can keep the nitrate

leached from cropped fields from entering local waterways. Native or planted perennial vegetation at streamsides can immobilize N in growing biomass and SOM (108). Waterway grass strips can also trap soil particles that would otherwise erode organic N into surface waters.

2. Restoring stream channels and wetlands in agricultural watersheds can promote denitrification and other microbial processes that convert nitrate to inert or less mobile forms of N (109). Channelization effectively turns headwater streams and wetlands into pipes that are less conducive to N retention (110) because water moves out faster and has less contact with soils, aquatic sediments, and wetlands, particularly during periods of high flow and nitrate export (111–113).

3. Specific fields or topographic positions can be targeted for intensive mitigation. It is increasingly clear that much nonpoint source pollution from agriculture arises from relatively small fractions of the landscape (e.g., Reference 114). Planting forage or other perennial crops, such as cellulosic biofuels (115), or restoring native species in these areas could reduce landscape-level N outputs. Restoring or enlarging wetlands in low-lying lands could even convert these areas from N sources to N sinks.

Better Management of Nitrogen in Animal Agriculture

Nitrogen is cycled relatively efficiently in pasture-based animal production systems: While the animals are pastured, excreted nitrogen is returned to the soil to be recycled into new forage. Properly managed, e.g., where overstocking and animal intrusion into streams are avoided (79), little N need be lost to the environment. In intensive animal operations, however, retaining N within the animal-plant system is more problematic because of the spatial disconnect between animals in CAFOs and the location in which their feed is grown, as described above.

Strategies for improving the nitrogen performance of confined animal production systems fall into two main categories—diet adjustments to reduce the amount of N excreted per unit of production and waste handling procedures that avoid the loss of excreted N to the environment.

Precision feeding offers the greatest opportunity for improving animal excretion efficiencies (116). Providing digestible protein and amino acids in concentrations that meet nutritional needs without excess can substantially reduce N excretion rates. In nonruminants, for example, diets with 2% to 6% less protein supplemented with specific amino acids can decrease N excretion by as much as 30% to 40% in poultry (117) and 20% to 50% in swine (118). In dairy cattle, feeding lactating cows individualized rations on the basis of milk production can decrease N excretion by 34% (119), and for both dairy and beef cattle, adjusting metabolizable protein components to optimize microbial fermentation in the rumen can likewise improve N efficiencies significantly (116). Phase feeding, in which rations are adjusted for animal age, can further decrease N excretion rates for all major classes of confined animals.

Waste storage and disposal is equally important for containing the N fed to confined animals. In the United States, over 90% of farms with confined animal units dispose the manure on farm (120). Improved management to reduce the environmental leakage of manure N can include the following (79, 120–121):

1. Storing manure in lagoons rather than in open-air mounds or pits creates anaerobic conditions that inhibit nitrate formation and thereby denitrification losses of N_2O. Covered lagoons offer the additional potential for capturing CH_4 for electricity generation.

2. Applying manure to fields only during or immediately prior to periods of active plant growth will help to ensure that its N is immobilized by the crop rather than

leached to groundwater as nitrate or lost to the atmosphere as N_2O.

3. Adjusting the amount of manure applied to fields to the amount that meets crop N needs—as described above for synthetic N application—will prevent overfertilization with its associated N losses; and

4. Injecting liquid manure below the soil surface rather than surface application will prevent ammonia loss.

Social Barriers to Achieving Nitrogen Efficiency

None of the potential practices for lowering N loading are cost free, and all involve changing human behavior. Motivating behavioral change requires more than making technology available and more even than making it affordable; we also need to understand the incentives and disincentives to which producers will be responsive. Potential incentives can include the following:

1. Green payments could be issued for ecosystem services valued by society that are provided by farmers (122). Services such as the wildlife diversity provided by conservation plantings, or clean water provided by better N stewardship, or biocontrol provided by more diverse rotations could be promoted by direct payments (as in the European Union) or by tax credits. Such payments or credits are the converse of crop subsidy payments based on production, which currently (and perversely) promote the overapplication of N fertilizers.

2. Carbon market credits could be created for N_2O mitigation. Providing carbon credits for avoided N_2O emissions could benefit N conservation in general because the same underlying process that ultimately controls N_2O fluxes—soil N availability—also controls the escape of other forms of reactive N. Evidence that N_2O emissions are disproportionately higher at N fertilization levels that exceed economically optimum N rates (123)

suggests that significant N_2O reduction could be achieved with little if any yield penalty (57).

3. Farmers could be protected from the perceived risk of underfertilizing. Many farmers use extra N fertilizer as a hedge against missing potential high yields in years with exceptionally favorable growing conditions. Because those conditions cannot be predicted, it can be rational to overfertilize in the hopes that "this year will be the one in ten" when extra N will pay off. Paying farmers for the yield difference between an overfertilized strip and the larger portion of the field fertilized at a median optimum N rate can keep overall N use lower without financial risk to growers and can also lower their N fertilizer costs. Evidence that economically optimum N rates are stable across multiple years (e.g., Reference 124) suggests that consistent underfertilization will be rare with this approach.

An alternative to these incentive-based approaches is direct regulation of excess N by specifying allowable levels of N additions, surplus N balances (nutrient additions relative to removals in crops), or losses of N to aquatic systems and/or to the atmosphere. The European Union has followed this path to a far greater extent than has the United States. For example, the United Kingdom undertook an extensive voluntary, compensated program (Nitrate Sensitive Areas) in 1989 to reduce the delivery of nitrate to surface waters (125). This program was superseded by a European Union program (Nitrate Protective Zones) beginning in 1998, which includes stricter requirements and sanctions for noncompliance (121). Similarly, the Netherlands undertook a compulsory national program based on the N balances of intensive agricultural systems; but in 2003, the European Court of Justice ruled that this program failed to meet the standards of the European Nitrates Directive and required the implementation of an alternative scheme that limits applications of N (126). Regulation

of emissions of N-containing trace gases in Europe may be developing along a similar path (127). Whether an approach is sought in incentives, regulations, or some combination of the two, it will be a substantial challenge to motivate people to make the necessary fundamental changes to agricultural enterprises in order to mitigate environmental damage from fertilizer N. A further challenge is that, in many regions, the long transit time of water and nitrate from farm fields to surface waters means that it will take decades before actions that reduce N losses from agricultural systems lead to substantial improvements in downstream water quality (125). It will be difficult to motivate people to take action now that will lead to improved water quality only in 30–50 years, but the alternative of pushing the problem beyond our children's time is untenable to many.

CONCLUSIONS

N excess has been recognized as an international problem linked to agriculture for at least 40 years (e.g., References 128–131). Its intractability is part technical and part social. There is no single technology or practice that can fix N excess, and as for other complex environmental issues of global importance, a portfolio approach is required: Some combination of appropriate strategies for utilizing N efficiently and guarding against unintentional loss to the environment, a social means for motivating the adoption of appropriate practices, and ongoing assessments of success.

Most N is applied to and lost from annual cropping systems. Targeting these systems for mitigation could provide measurable attenuation of reactive N loading. Four proven technologies are currently deployable.

- Increase rotational complexity with cover crops and greater diversity. Reducing the period of time that soil N is freely exposed to loss by microbial action, wetting fronts, and erosion can substantially reduce N losses during the nongrowing season. Cover crops can guard against

loss by immobilizing inorganic soil N in plant biomass and by protecting the soil from erosion and runoff. Likewise, mixing crops with high NUEs in a rotation with low-NUE crops, such as corn, can improve the N performance of the entire rotation cycle.

- Provide farmers with decision support tools that allow access to better fertilizer N recommendations. Estimates of economically optimum N rates can be improved with additional information about current soil and plant N status. Models can be used to extend estimates to fields for which empirical evidence is not available, and advanced crop sensors (e.g., Reference 132) provide a practical means for further adjusting N rates to current-season conditions.

- Use advanced fertilizer technology, such as delayed-release formulations and precision application methods. Farmers unable or unwilling to apply recommended rates of fertilizer to field crops after planting, when the crop is entering its period of maximum N uptake, might instead fertilize with a polymer- or sulfur-coated formulation that will release N during the appropriate period even when applied preplanting. Additionally, applying fertilizer in a geographically precise manner to a previously mapped field or in an on-the-go manner with advanced real-time sensors (105) will help to avoid overfertilizing those portions of the field with low-yield potentials and underfertilizing high-yield areas.

- Watersheds can be managed so that riparian areas, wetlands, streambeds, and other portions of the landscape that are downstream from agricultural fields can capture N that is lost from fields—either by retaining it within the watershed or by redirecting losses to less damaging pathways like denitrification to N_2.

In all, deployable solutions that would achieve high yields, use less N, and decrease N losses to the environment exist. Social

mechanisms that can encourage the adoption of these techniques exist as well. What remains most uncertain is society's willingness to pay for their implementation.

SUMMARY POINTS

1. Added nitrogen (N) is a crucial component of modern crop production, necessary to meet current and future food production needs.

2. A substantial proportion of the N added to cropping systems escapes to the environment.

3. The environmental cost of escaped N is substantial and includes coastal zone eutrophication, compromised drinking water and air quality, climate warming, and biodiversity change in receiving ecosystems.

4. The N cycle of cropping systems is complex but offers multiple points for management intervention.

5. Interventions currently available to improve N retention in row crops include diversifying primary and cover crops in current rotations, better matching fertilizer and irrigation inputs to crop needs, and better timing and precision placement of N fertilizer.

6. In animal agriculture, interventions available now include precision feeding and other dietary changes to reduce N excretion as well as better manure handling and application technologies.

7. N that escapes the farm can be at least partly mitigated by improved stream and wetland management.

8. There are currently few incentives for farmers to reduce environmental N loadings; potential incentives might include stewardship payments for ecosystem services, carbon market credits, and yield risk abatement.

FUTURE RESEARCH

The development of agricultural landscapes that are N conservative requires a systems-level research approach that is experimental, long-term, and geographically based.

At the local scale, a more complete and greater diversity of whole-farm N budgets is needed in order to better identify the most productive and cost-efficient intervention strategies for different management systems.

At the watershed and regional scales, we need long-term assessments of the factors that control the responsiveness of watersheds to management.

Four needs are particularly acute for addressing geographically scalable N questions: geospatial data to provide the ability to understand and predict N fluxes at landscape scales, sensor networks to quantify the impacts of management changes in different water- and airsheds, quantitative models to allow different management scenarios to be tested effectively, and long-term experiments and assessments.

Socioeconomic research to define the most cost-effective incentive (or disincentive) framework for promoting the use of N conservation strategies.

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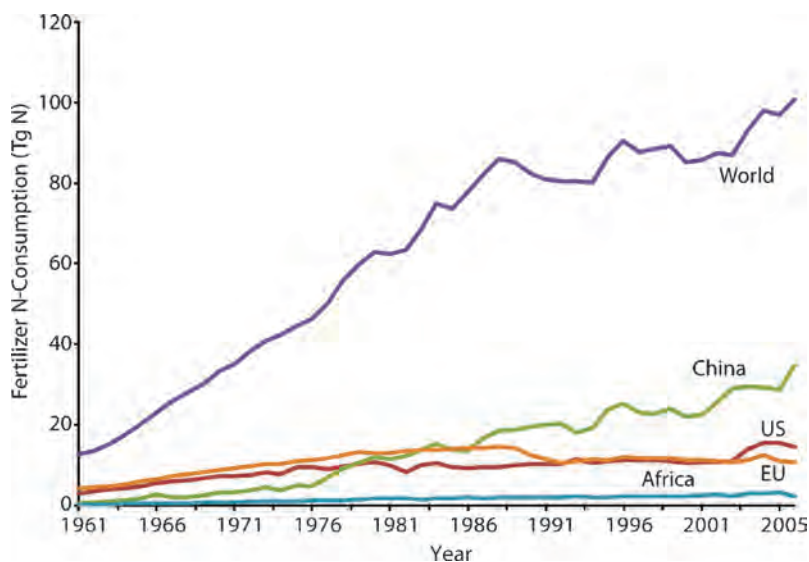135. Jackson L, ed. 1997. *Ecology in Agriculture*. New York: Academic

Figure 1

Nitrogen (N) fertilizer consumption by region and global production, 1961–2006 (133). Abbreviatons: EU, European Union; Tg, one million tonnes; US, United States.
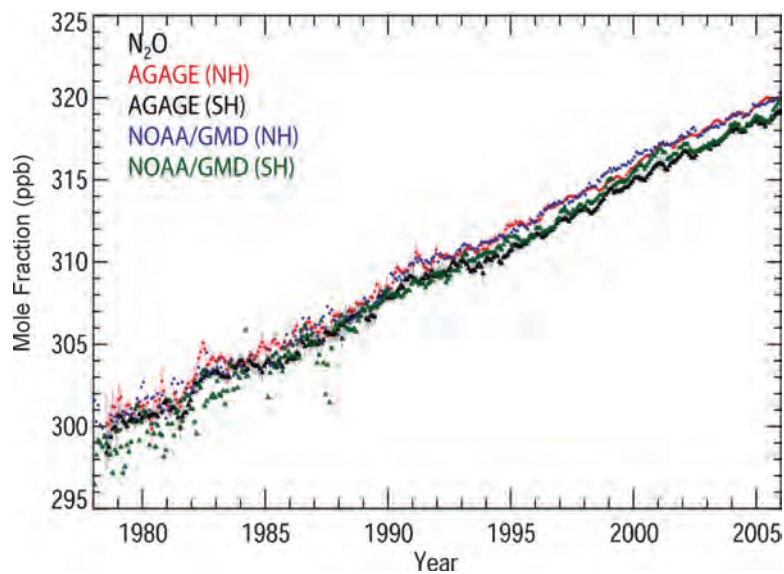
Figure 3

Atmospheric nitrous oxide (N_2O) concentrations, from 1980 through 2005 (26). Abbreviations: AGAGE, Advanced Global Atmospheric Gases Experiment; NH, Northern Hemisphere; SH, Southern Hemisphere; NOAA/GMD, National Oceanic and Atmospheric Administration Global Monitoring Division; ppb, parts per billion.
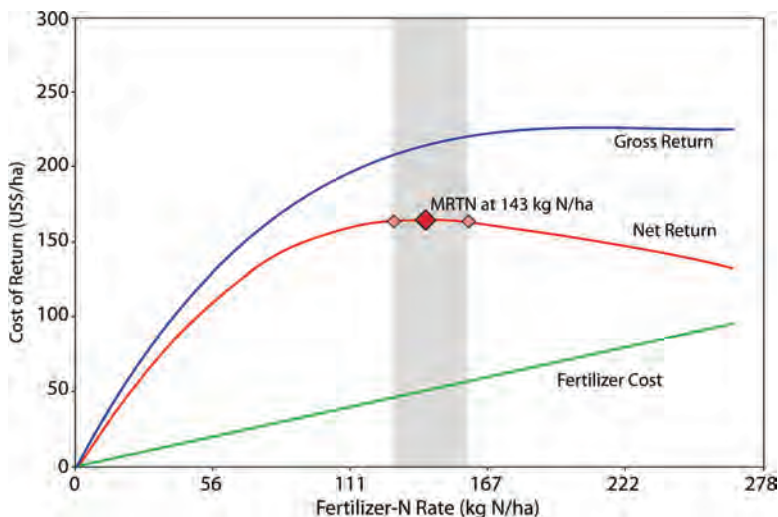
Figure 5

Maximum nitrogen (N) return for an Iowa corn crop following soybean. Results based on 165 sites for an N price of US$880/MT ($0.40/lb) and a corn price of US$157/MT ($4.00/bu). The mean return to N (MRTN) rate of 143 kg N/ha (128 lb N/acre) provides 98% of maximum yield, which occurs at ~196 kg N/ha (175 lb N/acre) (134).

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Errata

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