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Nitrogen in crop production: An account of global flows

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Abstract. Human activities have roughly doubled the amount of reactive N that enters the element's biospheric cycle. Crop production is by far the single largest cause of this anthropogenic alteration. Inorganic fertilizers now provide 80 Tg N yr^{-1} (Tg = 10^{12} g), managed (symbiotic) biofixation adds about 20 Tg N yr^{-1}, and between 28 and 36 Tg N yr^{-1} are recycled in organic wastes. Anthropogenic inputs (including N in seeds and irrigation water) now supply about 85% of 170 (151-186) Tg N reaching the world's cropland every year. About half of this input, 85 Tg N yr^{-1}, is taken up by harvested crops and their residues. Quantification of N losses from crop fields is beset by major uncertainties. Losses to the atmosphere (denitrification and volatilization) amount to 26-60 Tg N yr^{-1}, while waters receive (from leaching and erosion) 32-45 Tg N yr^{-1}. These N losses are the major reason behind the growing concerns about the enrichment of the biosphere with reactive N. The best evidence suggests that in spite of some significant local and regional losses, the world's agricultural land accumulates N. The addition of 3-4 billion people before the year 2050 will require further substantial increases of N input in cropping, but a large share of this demand can come from improved efficiency of N fertilizer use.

1. Introduction

Human interference in the global nitrogen cycle has become a topic of increasing research attention [*Smil*, 1990, 1991, 1997a; *Kinzig and Socolow*, 1994; *Galloway et al.*, 1995; *Jordan and Weller*, 1996; *Vitousek et al.*, 1997]. Compared to the preindustrial era, human activities have roughly doubled the amount of reactive N that enters the element's biospheric cycle [*Galloway et al.*, 1995; *Smil*, 1997a]. These compounds include NO_x (NO and NO_2) and other oxygenated single-N species (the two categories jointly designated as NO_y), NH_3 and NH_4^+ (labeled NH_x), and organic N, and there has been also an increase in concentrations of N_2O, a greenhouse gas. Effects of this N enrichment range from atmospheric changes (higher concentrations of NO_x, NO_3^-, and NH_x) to alterations of terrestrial and aquatic ecosystems (caused by acidification, nutrient leaching, and eutrophication) and to impacts on carbon cycle.

Crop production is by far the single largest cause of human alteration of the global N cycle: *Rhizobium* bacteria symbiotic with leguminous crops fix much more N than would be the case if natural plant communities were occupying the same space, and N applied to fields in synthetic fertilizers (now about 80 Tg N yr^{-1}) is about 4 times as large as is the total amount of the element that humans fix by burning all fossil fuels [*Galloway et al.*, 1995; *Smil*, 1997a].

This paper accounts first for nitrogen taken up by the global harvest of agricultural crops and their residues, and then it reviews all major natural and anthropogenic flows of N reaching, and leaving, the world's crop fields. Quantified inputs include nitrogen in atmospheric deposition, seeds, irrigation water, recycled crop residues, animal manures, inorganic fertilizers, and the nutrient supplied by biofixation; outputs include, besides the harvested biomass, the losses due to gaseous emissions of NO, N_2 and N_2O, volatilization of NH_3, leaching, soil erosion, and losses from tops of plants. In conclusion I evaluate the N balance of the global cropping and offer a brief assessment of ways to reduce the existing N burden.

Many quantifications in this paper (i.e., nitrogen removed from fields in crops and in crop residues and nitrogen lost in leaching and in eroded soils) are my original calculations based on the best available statistics and on other relevant information detailed in the text; others are derived from a wide-ranging survey of rapidly increasing literature on nitrogen cycling. Because of our poor understanding of many fluxes of the complex nitrogen cycle and because of considerable spatial, as well as secular, variation of most of the flows, quantifications of global N fluxes should avoid an appearance of unwarranted accuracy. Consequently, I will present all inputs and losses (Tables 5 and 6) as ranges rather than as single values.

2. Global Crop Harvest

The Food and Agriculture Organization's (FAO) worldwide database is the most comprehensive source of statistics on arable land, annual crop production, and yields (see FAOSTAT Agriculture Data available as http://apps.fao.org, hereinafter referred to as FAOSTAT Data). Accuracy of these figures is high (errors mostly smaller than 5%) for high-income nations, but the FAO must make many in-house estimates to fill numerous data gaps for low-income countries. FAO statistics also do not take into account home gardens and backyard plots, which make important nutritional contributions in most Asian, African, and Latin American countries [*Hoogerbrugge and Fresco*, 1993]. Moreover, some official farmland figures submitted by the FAO's member states are known to be substantial (more than 10%) underestimates. Perhaps the most notable example of such underestimates is the case of China's farmland: Its real total is

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50% larger than the official claim, at least 140 Mha rather than 95 Mha [*State Statistical Bureau, 1997; National Intelligence Council,* 1997].

Consequently, the actual area of globally cultivated land is almost certainly larger than the FAO total of 1.477 Gha (1.362 Gha of arable land and 115 Mha of land under permanent crops) and so is the aggregate crop harvest. Increases of up to 5% appear to be highly plausible, but the absence of detailed information needed to quantify and to apportion these additional amounts of farmland and harvests precludes any satisfactory corrections of FAO's statistics even on a continental basis. According to uncorrected FAO data, the mid-1990s global crop harvest, taken as the mean of 3 years (1994-1996) and excluding the forages grown on farmland, averaged almost 5.5 Pg yr^{-1} in fresh weight (Table 1).

The most accurate way to calculate N taken up by this output is to use N contents for harvested parts (grains, stalks, leaves, roots, and fruits) of every major crop. I have done this calculation for 40 separate crops aggregated into 8 principal categories. Because N content of crops is commonly expressed in terms of absolutely dry matter, appropriate conversions must be done first. I used average moisture and N values from *National Research Council (NRC)* [1972] and *Bath et al.* [1997]. The mid-1990s global crop harvest of about 2.65 Pg yr^{-1} of absolutely dry matter contained 50 Tg N yr^{-1} (Table 2). Cereals accounted for 60% of total dry crop mass, and they also contained 60% of all removed N; sugar crops ranked second in dry mass (16%), but legume crops were second in assimilated N (20%).

Calculations of N incorporated by crop residues cannot be done with a comparable accuracy as no country keeps statistics on their production. Published estimates of national or global totals of crop residue production were prepared in order to evaluate options for agronomic management and animal feeding or to find potential contributions of biomass energies or total emissions of greenhouse gases [*United States Department of Agriculture (USDA),* 1978; *Smil,* 1985, 1994; *Kossila,* 1985; *Andreae,* 1991]. Many figures are available for harvest indices (HI) of major crops, the ratios of crop yield and total above-ground phytomass [*Donald and Hamblin,* 1976; *Gifford and Evans,* 1981; *Hay,* 1995]. Residual biomass expressed as a multiple of harvested yields can be obtained simply as a ratio of (1-HI)/HI, but this total will be somewhat larger than the commonly used residue/crop (R/C) ratios as the latter indices may not include plant stubble.

Table 1. Annual Global Harvest of Crops and Crop Residues in the Mid-1990s

Crops	Harvested Crops Tg of fresh weight	Harvested Crops Tg of dry weight	Crop Residues Tg of dry weight	Total Harvest Tg of dry weight
Cereals	1900	1670	2500	4170
Sugar crops	1450	450	350	800
Roots, tubers	650	130	200	330
Vegetables	600	60	100	160
Fruits	400	60	100	160
Legumes	200	190	200	390
Oil crops	150	110	100	210
Other crops	100	80	200	280
Forages	2500	500		500
Total	7950	3250	3750	7000

Fresh weights of harvested crops are according to FAOSTAT Data; dry weight is calculated by using standard moisture contents by *Watt and Merrill* [1975] and *Bath et al.* [1997]; derivation of crop residue totals are described in the text. All figures are rounded to avoid the appearance of unwarranted precision.

Table 2. Nitrogen Incorporated in the World Crop Harvest of the Mid-1990s

Crops	Harvested Crops	Crop Residues	Total Harvest
Cereals	30	15	45
Legumes	10	5	15
Sugar crops	2	1	3
Roots, tubers	2	1	3
Vegetables, fruits	2	1	3
Other crops	4	2	6
Forages	10	...	10
Total	60	25	85

All figures are in Tg N yr^{-1}. Fresh weights of crops and dry weights of crop residues from Table 1 are multiplied by average N contents given by *NRC* [1972], *Watt and Merrill* [1975], *Smil* [1994], and *Bath et al.* [1997].

The standard practice is to quote the residue in terms of dry-matter mass and the crop yield at field moisture. The R/C ratios vary among cultivars as well as for the same cultivar grown in different environments. Agronomic factors (planting date, irrigation regime, and N applications) can also make substantial differences [*Prihar and Stewart,* 1991; *Roberts et al.,* 1993]. Assumptions concerning average straw/grain (S/G) ratio of cereals will make a particularly large difference in global calculations. Using average S/G ratio of 1.3, rather than 1.2, adds almost 200 Tg more straw, a total larger than all residual phytomass produced by the world's tuber and root crops. I tried to minimize such errors by calculating residue output separately for 40 different crops rather than just for major crop categories; still, the grand total could not be calculated with an error smaller than 5%.

I used FAO's crop production figures, standard values for water content of harvested parts, and fairly conservative R/C multipliers [*Smil,* 1994, 1999]. These calculations result in outputs of between 3.5 and 4.0 Pg yr^{-1} of crop residues during the mid-1990s; the most likely total of 3.75 Pg is about 1.4 times the aggregate crop harvest (Table 1). Cereal straws, stalks, and leaves accounted for 2/3 of all residual phytomass, and sugar cane tops and leaves were the second largest contributor. Just over 60% of all residual phytomass were produced in low-income countries, and close to 45% of it were originated in the tropics. Substantial interspecific and intraspecific variations result in a rather broad range of N contents of cereal straws (0.4-1.3%, values around 0.6% are common); only leguminous straws are relatively N-rich. The total removed in residues amounted to about 25 Tg N yr^{-1} during the mid-1990s or 1/3 of the total taken up by crop biomass (Table 2).

Quantification of annual harvests of forages grown on arable land (alfalfa, clovers, vetches, and various legume-grass mixtures) can be done only approximately. These crops include all-legume or legume-grass swards cultivated intermittently on arable land, either for direct foraging or, more often, for roughage feed (hay and silage), as well as a variety of green manures, i.e., leguminous species which are plowed under after 40-80 days of growth. FAO does not keep track of these crops, but there is no doubt that their worldwide extent has been declining: diffusion of intensive grain monocultures and wider use of synthetic fertilizers have been the main reasons for these reductions [*Smil*, 1993; *Wedin and Klopfenstein*, 1995].

I have consulted statistical or agricultural yearbooks for about 20 of the world's largest countries in order to derive the most likely range of the land under forages grown on cropland: between 100-120 Mha in the mid-1990s, with green manures accounting for no more than 20% of the total area. Fairly conservative assumptions about mean dry-matter harvests, 5 Mg ha^{-1} for alfalfa (planted on about 30% of cropland pastures) and 4 Mg ha^{-1} for other leguminous and mixed legume-grass forages, result in the total yield of about 500 Tg containing 10 Tg N. This would increase the mid-1990s global dry-matter crop harvest to about 7 Pg yr^{-1}, with about 85 Tg N yr^{-1} incorporated in this phytomass (Tables 1 and 2).

3. Nitrogen Inputs

A small part of harvested N returns to fields in planted seeds and tubers. All fields receive N in dry and wet deposition of airborne compounds and in recycled roots and stubble. Irrigated fields receive additional N, largely from nitrates dissolved in water. More intensive recycling of straws and stalks is common in many agroecosystems and in all regions, with mixed farming animal manures produced in confinement also recycled. Biofixation contributes relatively small amounts of N in dry fields planted to nonleguminous crops, but it is a major source of the nutrient in leguminous cultures and in rice fields. Although many fields, especially in Africa, have never received any inorganic fertilizer, synthetic N compounds are now the single largest input of the nutrient in the world's crop production.

3.1. Planted Seeds and Tubers

This is a minor input, which is fairly easy to quantify. Many agricultural data sources carry information on seeding rates, mostly in terms of mass units per area, sometimes as shares of harvested crops. Mean seeding rates for more than a dozen principal crops are listed in detailed food balance sheets prepared by the FAOSTAT Data and multiplied by respective planted areas tabulated in the FAOSTAT Data. In turn, these products were multiplied by average N contents of seeds and tubers used to calculate N incorporated in the global crop harvest (Table 2). This procedure resulted in an annual return of about 2 Tg N in seeds and tubers.

3.2. Atmospheric Deposition

Higher production of nitrates (generated mainly by oxidation of NO_x released by combustion of fossil fuels) and volatilization of NH_3 (from animal wastes, soils receiving ammoniacal fertilizers, and plant tops) have been the main causes of steadily increasing wet and dry deposition of N compounds on agricultural land. Because of the brief atmospheric residence time of ammonia compounds, large shares of NH_x originating from agricultural activities either fall back on the agricultural land from which they originated or are deposited close to the areas of their emissions; in contrast, NO_y compounds are carried much further downwind before they are precipitated or deposited in dry form. Dry and unpolluted regions have the lowest rates of N deposition, while the annual inputs in the most intensively farmed, industrialized and densely populated areas of North America, Europe and Asia are commonly an order of magnitude higher (see the National Atmospheric Deposition Program's data available as http://nadp.nrel.colostate.edu/NADP, hereinafter referred to as NADP Data, and *Erisman* [1995]).

U.S. regional means of N deposited in precipitation are an excellent illustration of these differences (see NADP Data). Most of the country west of the Mississippi annually receives just 1-2 kg N ha^{-1} in wet deposition, with NH^{4+} inputs negligible nearly everywhere west of the Rocky Mountains, while total N in precipitation averages over 7 kg N ha^{-1} in the coastal Northeast, and it would be much higher if as much 75% of the area's NO_x emissions were not exported out to the Atlantic Ocean [*Jaworski et al.*, 1997]. Europe has even greater regional differences: At over 20 kg N ha^{-1}, the annual rates of NH_4^+ in precipitation in the Netherlands, northeastern France, and southern England are more than 20 times as high as those in southern Spain or Italy [*Van Leeuwen et al.*, 1996].

I used separate estimates of continental averages of wet NO_y and NH_x deposition rates, derived from long-term deposition measurements (see NADP Data and *Erisman* [1995]), from emission inventories [*Bouwman et al.*, 1997] and from approximate global balances [*Berner and Berner*, 1996], and FAOSTAT Data's farmland statistics to calculate the total input of about 13 (11-15) Tg N yr^{-1}. As we have no comparatively widespread and reliable measurements of dry deposition, all generalizations are highly uncertain. Minimum estimates can be derived by assuming that dry NH_x deposition is equal to at least 1/3 of the wet flux and that dry NO_y is equal to at least 3/4 of the wet input [*Warneck*, 1988]. The actual rates, particularly for dry NH_x deposition, may be much higher [*Sutton et al.*, 1993].

These adjustments would result in total deposition of about 20 (18-22) Tg N on the world's agricultural land. As the total terrestrial deposition of NO_y and NH_x is now about 60 Tg N yr^{-1} [*Galloway et al.*, 1995; *Berner and Berner*, 1996], this would mean that at least 1/3 of all reactive N deposited on land settles on the farmland which accounts only for some 11% of ice-free surfaces. This higher level of enrichment is expected owing to a close proximity of agricultural regions and nonagricultural sources of NO_x emissions (particularly in Europe, eastern North America, and east Asia) and to a large share of NO_x emissions emanating from agricultural activities.

3.3. Irrigation Water

About 17% of the world's cropland (250 Mha) were irrigated in 1995, with nearly 2/3 of the global total in Asia (FAOSTAT Data). Irrigation waters always contain some N, and its concentrations are relatively high (more than 5 mg N kg^{-1}) in all intensively cultivated regions (where the leaching of N from fertilizers and animal wastes are the main sources) as well as in densely populated and heavily industrialized areas (where human wastes and industrial pollutants are the main contributors).

Annual inputs of the nutrient in irrigation water may be higher than 20 kg N ha^{-1} in double-cropped rice [*Wetselaar et al.*, 1981], or they may be below 5 kg N ha^{-1} in those predominantly rain-fed crops that receive only some supplementary irrigation. With average water applications of at least 9000-10,000 m^3 ha^{-1}

[*Shiklomanov*, 1993; *Postel et al.*, 1996] and with N concentrations mostly between 1-2 mg N kg^{-1}, the annual global inputs of the nutrient would be about 4 (3-5) Tg N.

3.4. Recycling of Crop Residues

Recycling of crop residues, directly by leaving them after the harvest to decay on field surfaces or by incorporating them into soil by ploughing, discing, or chiselling, and indirectly by using them in mulches and composts or returning them to fields in animal wastes, has been practiced vigorously by every traditional agriculture, and it remains an essential part of modern field management [*Smil*, 1999]. Besides the nutrient recycling, protection against water and wind erosion, enhanced water storage capacity of soils, and their enrichment with organic matter are other principal benefits [*Barreveld*, 1989].

Crop residues have many competing uses: They have been an important source of household fuel and building material in many low-income countries; they provide indispensable bedding and feed for animals, particularly ruminants, of all continents; they offer an excellent substrate for cultivation of mushrooms; and they have been used for making paper and as feedstocks for extracting organic compounds. Unfortunately, a significant share of crop residues is still burnt in fields, an agronomically undesirable practice which also generates greenhouse, and other trace, gases [*Andreae*, 1991].

No country keeps comprehensive statistics of crop residue uses. Fairly reliable information on major uses is available for some affluent countries, but elsewhere we have to rely on fragmentary data and expert estimates. *Intergovernmental Panel on Climate Change and Organization for Economic Cooperation and Development* (*IPCC and OECD*) [1997] estimated that about 25% of all residues are burnt in low-income countries and that the corresponding share in affluent nations is just 10%. The first rate is almost certainly too conservative, especially when one includes the use of residues for fuel: Widespread energy shortages in deforested rural areas make the residues the only accessible household fuel for hundreds of millions of poor peasants. Chinese estimates indicate that at least 3/5 of all crop residues are burnt by rural households [*Smil*, 1993; *Sinton*, 1996].

The estimated mean for residue burning in high-income countries is also most likely too low, as some fairly reliable data on average burn fractions indicate regionally much higher rates both for field and orchard crops [*Jenkins et al.*, 1992]. Adding up field and household combustion, a more realistic assumption would be that 35% of all residues in low-income countries and 15% of all residues in affluent nations are burnt. Burning of about 1 Pg of crop residues containing roughly 6 Tg N would release, assuming 80% combustion efficiency, almost 5 Tg N. These emissions would not be made up only of NO_x and NH_3: 30-40% of the element present in the phytomass is converted during flaming combustion directly into N_2 [*Kuhlbusch et al.*, 1991].

Feed and bedding account for anywhere between 20-25% of the remaining total in high-income nations and for up to 1/3 in low-income countries with large numbers of domestic animals. Other uses are negligible, particularly when compared to inherent errors in estimating the production of crop residues in the first place. Consequently, the highest plausible recycling rates, about 70% in high-income and 60% in low-income countries, would return annually about 16 Tg N to the world's croplands. The lowest plausible estimates, with average recycling rates of 60% in high-income and 40% in low-income countries, would return about 12 Tg N yr^{-1}.

3.5. Animal Manures

Manure production, and its N content, depends on breed, sex, age, health, feeding, and water intake of animals [*Smil*, 1985]. Fresh waste output per head can be estimated fairly reliably as a share of animal's live weight, but large variations in body mass and feeding, especially the differences between confined and well-fed animals in modern settings and traditional, free-ranging and poorly fed (and often still also hard-working) breeds, make even national means uncertain.

Choosing the best means of N content of manures is even more difficult as various reports list twofold to threefold ranges for dairy, pig, and poultry manures [*Misra and Hesse,* 1983; *Whitehead and Raistrick,* 1993; *Choudhary et al.,* 1996]. *Powers et al.* [1975] found an even greater range (0.6-4.9% N) for beef manures. Using the means listed in Table 3 and multiplying them by the FAOSTAT Data animal head counts of the mid-1990s results in annual voiding of about 75 (70-80) Tg N. This is less than several other recently published estimates by *Mosier et al.* [1998]; *Bouwman et al.* [1997] and *Nevison et al.* [1996].

The main reason for this difference is that I am assuming lower averages of body weights and poorer feeds for cattle, sheep, and goats in low-income countries (they now have about 75% of the world's bovines and almost 80% of all sheep and goats). Even so, my estimate for nitrogen in cattle manure may be still be too large as some source credit Indian cattle manure with just 0.7% N on a dry-weight basis [*Singh and Balasubramanian*, 1980].

Quantifying the amount of N returned to cropland is even more uncertain because of the need for concatenated assumptions. Only animals grazing on cropland pastures, harvested fields, or cover crops in tree plantations will deposit their wastes directly on the farmland: This contribution is only a small share of all N recycled in manure. Wastes produced by

Table 3. Average Annual Production of Animal Waste Solids and Their Typical Nitrogen Content

Animals	Waste Solids, kg $head^{-1}$	N, %	N Output, kg $head^{-1}$	Manure Produced in Confinement, %	NH_3-N Losses, %
Dairy cattle					
Modern	2000	4	80	60	36
Traditional	1200	3	45	65	36
Nondairy cattle					
Modern	1200	4	50	35	36
Traditional	900	3	30	25	36
Water buffalo	900	3	30	35	28
Horses, mules	1200	3	35	50	28
Pigs	300	3	10	100	36
Sheep, goats	150	3	5	10	28
Poultry	8	4	0.3	100	36

The following average live weights are assumed: modern dairy cattle, 500 kg; modern nondairy cattle and horses, 400 kg; traditional dairy and nondairy cattle and water buffaloes, 300 kg; pigs, 100 kg; sheep and goats, 40 kg; and poultry, 1.5 kg [*Misra and Hesse,* 1983, *Smil,* 1983, *Nordstedt,* 1992, and *Bouwman et al.,* 1997].

animals grazing on permanent pastures will be unavailable for recycling to cropland; only manure produced in confinement on mixed farms and in feeding facilities located in farming areas can be economically distributed to nearby fields.

Recycling rates range from only about 30% in the USA, where about 40% of all animal wastes are voided in confinement and about 75% of this output are actually returned to fields [*USDA*, 1978], to more than 90% in several small European countries [*Rulkens and ten Have,* 1994]. Even greater differences are seen in Asia: Hardly any manure is used as fertilizer in the continent's interior; a large share of cattle manure in the Indian subcontinent is gathered for fuel, while about 80% of China's pig manure is recycled.

Using average shares of manure produced in confinement (Table 3) that are nearly identical to those of *Bouwman et al.* [1997], I calculated the total amount of N voided annually in stables, barns, sheds, and corrals at 30 (25-35) Tg N or about 40% of the nutrient in all animal manures. Even if this manure were to be completely recycled its initial N content would be greatly diminished before these organic wastes could enrich agricultural soils: with more than 70% of urine N voided as urea, there are large losses due to rapid hydrolysis of the compound and subsequent volatilization of NH_3 during collection, storage, composting, and handling of wastes [*Subba Rao,* 1988; *Eghball et al.,* 1997; *Bouwman et al*., 1997].

These losses, together with denitrification and leaching, are especially high in traditional settings with prolonged storage of manures in the open. Depending on the kind of handling and storage, most or nearly all of NH_3-N (up to 50% of initially voided N), as well as appreciable amounts of NO_3-N and organic N, may be lost before the application to fields. Assuming that 90% of manure produced in confinement is eventually recycled to fields (the rest is dumped or recycled to pastures) and reducing its initial N content by average NH_3-N losses proposed by *Bouwman et al.* [1997] and listed in Table 3, I calculated that recycled animal manures contribute about 18 Tg N (14-22) Tg N. I will estimate specific post-application NH_3 losses in section 4.

3.6. Biofixation

Reduction of atmospheric N_2 to NH_3 is performed enzymatically by at least 60 genera of cyanobacteria (blue-green algae), 15 genera of symbiotic actinomycetes, and, above all, some 25 genera of free-living and symbiotic bacteria. By far, the most important symbionts belong to the genus *Rhizobium* associated with leguminous plants. Nearly a generation ago, *LaRue and Patterson* [1981] noted that there was not a single legume crop for which we had valid estimates of N fixation. Although many new estimates have been published since that time, we are still unable to offer reliable, representative values of average annual fixation rates even for the most important leguminous cultivars. This is not surprising when considering the large natural variability of symbiotic fixation, as well as the errors inherent in techniques commonly used to measure the rates of N fixation [*Hardarson and Danso*, 1993; *Danso et al.,* 1993].

Fixation rates vary enormously with both abundance and persistence of specific *Rhizobium* strains in soils, with the vigor of host plants and with environmental stresses [*Ayanaba and Dart,* 1977; *Broughton and Puhler*, 1986]. Major factors reducing, or even inhibiting, N fixation include soil temperature, both too little and too much moisture, low soil O_2 levels, high concentrations of inorganic forms of N, shortages of essential micronutrients (particularly Mo, V, and Fe present in the redox center of nitrogenase [*Eady*, 1995]), and low soil pH.

Table 4. Ranges of Published Biofixation Estimates and the Mean Values Used for Calculating Global Biofixation by Major Leguminous Species

Crops	Ranges of Published Estimates*, kg N ha^{-1}	Ranges of Means Used in Calculations, kg N ha^{-1}
Seed legumes		
Beans	3-160	30-40-50
Broad beans	45-300	80-100-120
Chickpeas	3-141	40-50-60
Lentils	10-192	30-40-50
Peas	10-244	30-40-50
Peanuts	37-206	60-80-100
Soybeans	15-450	60-80-100
Other pulses	7-235	40-60-80
Forages		
Alfalfa	65-600	150-200-250
Clovers	28-300	130-150-170
Other forages	9-180	80-100-120

* Data are from *LaRue and Patterson* [1981], *Heichel* [1987], *Giller and Wilson* [1991], *Peoples et al.* [1995], and *Smil* [1997b]

Dozens of N fixation rates from a number of different agroecosystems have been published for such widely cultivated leguminous food and feed crops as common beans (*Phaseolus vulgaris)* and soybeans (*Glycine max*) as well as for such leading forage species as alfalfa (*Medicago*) and various clovers (*Trifolium*). Almost all published values have at least three-fold ranges and much larger differences are common (Table 4). Particularly notable are low-fixation rates in common beans in Latin America, the crop's region of origin [*Henson*, 1993; *Tsai et al.,* 1993]. For less common leguminous species, we have too few reliable figures; our knowledge of fixation rates is particularly inadequate as far as tropical legumes, be they used for food, fodder, or as green manure, are concerned [*Blair et al.,* 1990; *Peoples and Herridge*, 1990].

In order to make the best possible estimate of symbiotic biofixation, I have used specific ranges for eight different kinds of seed legumes listed in Table 4 (beans, broad beans, chickpeas, lentils, peas, peanuts, soybeans, and other pulses) whose cultivated areas were taken from FAOSTAT Data. This resulted in the most likely total of global *Rhizobium* fixation of 10 Tg N yr^{-1} (range of 8-12 Tg N yr^{-1}), while leguminous crops removed 15 Tg N yr^{-1} (Table 2). This means that these crops derived 2/3 of their N from biofixation, an average share in excellent agreement with numerous studies of biofixation's contribution to N requirements of legumes [*Hardarson*, 1993; *Peoples et al.*, 1995].

This does not mean that there is no transfer of N from seed legumes to subsequent nonleguminous crops. This eventual enrichment depends on the degree to which legumes can satisfy their total N needs from the biofixation and on the share of the fixed N taken up by the harvested legume seeds [*Hardarson*, 1993; *Giller et al.*, 1994]. The latter share ranges from as little as 30% for common beans to more than 80% for soybeans.

Complete recycling of bean plant residues will thus transfer most of the fixed N for eventual use by subsequent crops; however, soybeans, although they are much more prolific N fixers than beans, may not be able to provide all of the needed nutrients, and the crop will have to claim considerable amounts of soil N [*Heichel*, 1987].

Estimating biofixation by leguminous cover crops is a much greater challenge. As already noted, forages and green manures occupy 100-120 Mha of farmland. Average fixation rates of 200 kg N ha^{-1} for alfalfa and 150 kg N ha^{-1} for clovers and vetches are well supported by numerous measurements [*Frame and Newbould*, 1986; *Dovrat*, 1993; *Meelu et al.*, 1994], but fixation by legume-grass mixtures is much more variable, ranging from a just a few kilograms to more than 250 kg N ha^{-1} [*Heichel and Henjum*, 1991; *Farnham and George*, 1994]; I will assume just 50 kg N ha^{-1}. A fairly conservative total of N fixation in 100-120 Mha of cropland forages and green manures (averaging 100 kg N ha^{-1}) is then about 12 Tg N yr^{-1} (range of 10-14 Tg N yr^{-1}).

As these crops remove annually only about 10 Tg N, there can be no doubt that they leave behind large amounts of N for subsequent nonleguminous crops. The actual total is almost certainly somewhat higher as it includes biofixation by leguminous crops grown as soil covers in many tree plantations as well fixation by trees and shrubs used in alley cropping and planted along field boundaries and house gardens for feed, fuelwood, and shade [*Blair et al.*, 1990; *Gutteridge and Shelton*, 1994]. Biofixation by non-*Rhizobium* diazotrophs is of lesser importance. Some studies report values for biofixation by free-living bacteria and cyanobacteria in cereal fields in humid environments in excess of 20 kg N ha^{-1} during the growing season [*Neyra and Dobereiner*, 1977], but typical rates in drier environments are mostly less than 5 kg N ha^{-1} or no more than 4 (2-6) Tg N yr^{-1} for the world's cereal, tuber, and oil crop fields.

In contrast, cyanobacteria (mainly *Anabaena* and *Nostoc*) in rice fields can fix 20-30 kg N ha^{-1} during the growing season. When including the contributions by *Anabaena azollae*, a symbiont of *Azolla pinnata*, a small floating fern cultivated in Asia's paddies, which can produce as much as 50-90 kg N ha^{-1} during 40-60 days [*Meelu et al*, 1994], and when taking into account the rate of rice multicropping (mean global ratio of about 1.25), cyanobacterial fixation may have contributed between 4-6 Tg N yr^{-1} in 150 Mha of the world's paddies. A recent discovery of endophytic diazotrophs (*Acetobacter* and *Herbaspirillum*) living inside sugar cane roots, stems, and leaves explains why unfertilized crops maintain high yields even after years of consecutive cultivation [*Boddey et al.*, 1995]. These endophytes fix annually at least 50 kg N ha^{-1}, maxima may be higher than 150 kg N ha^{-1}, resulting in the range of 1-3 Tg N for the world's 18 Mha of sugar cane.

As calculated above, the total biofixation in crop fields, plantations, and cropland pastures, a combination of contributions by *Rhizobium*-legume symbioses in food, feed, forage, and green manure crops; symbiotic *Azolla-Anabaena* fixation in paddies; free-living cyanobacteria and bacteria in wet and dry fields; and endophytic bacteria in sugar cane, would have been 33 (25-41) Tg N yr^{-1} during the mid-1990s. This was less than half the total mass of N applied in inorganic N fertilizers, but more than all N recycled in crop residues and animal manures.

3.7. Inorganic Fertilizers

This is the most accurately known, and now also the single largest, input into the global N cycle. Mineral compounds (KNO_3 and $NaNO_3$) are still used, but their applications are dwarfed by the use of synthetic fertilizers. Their production now begins invariably with the Haber-Bosch process of NH_3 synthesis first introduced commercially in 1913 [*United Nations Industrial Development Organization and International Fertilizer Development Center* (*UNIDO and IFDC*), 1998]. During the mid-1990s the nominal capacity of the world's ammonia plants, now using largely CH_4 both as the feedstock (H source; N is separated from the air) and fuel, was about 115 Tg N yr^{-1} [*Food and Agriculture Organization* (*FAO*), 1997]. Actual synthesis was about 95 Tg N yr^{-1}, of which almost 80 Tg N yr^{-1} was incorporated in synthetic fertilizers with the remainder consumed by chemical industries and lost during processing and transportation.

Consumption data, global, regional, and national, are reported annually by the FAOSTAT Data as well as by the International Fertilizer Association (see the World Nitrogen Fertilizer Consumption report available as http://www.fertilizer.org). After a slight decline during the early 1990s (to 72.5 Tg N by 1993), global consumption rose again to almost 79 Tg N in 1995 and to nearly 83 Tg N in 1996. These consumption figures do not account for a variety of preapplication losses which are particularly large for the highly volatile NH_4HCO_3, still a major fertilizer in China [*Smil*, 1993]. In addition, some N fertilizers are used on permanent pastures and in forests, and increasing quantities are also applied to lawns.

Ammonia is either used directly as the most highly concentrated (82% N) and the cheapest N fertilizer (applied as gas or in aqueous solutions) or converted to urea, the most concentrated solid fertilizer (46% N) now accounting for nearly 40% of global N consumption and greatly preferred in Asia, the continent which now uses nearly 3/5 of the world's fertilizer N. Various nitrates or compound fertilizers (combinations with P, K, and micronutrients) are also used. About 60% of synthetic N fertilizers are now consumed in the low- and middle-income countries, and the nutrient is indispensable for maintaining the recently achieved staple food self-sufficiency in the world's most populous nations (China, India, and Indonesia).

Annual application means of the mid-1990s were around 50 kg N ha^{-1} of arable land for the world, and they ranged between 10-100 kg N ha^{-1} for continents (Africa and Europe being the extremes). National application means hide enormous regional variations: For example, the U.S. mean of about 50 kg N ha^{-1} is made up of rates below 20 kg N ha^{-1} in North Dakota wheat fields to well over 200 kg N ha^{-1} in Iowa corn fields. Application rates should be also adjusted for various degrees of multicropping: High fertilization rates in Asia's monsoonal agricultures, where double- and even triple-cropping is common, are then halved or cut by 2/3 in order to make proper comparisons with those temperate countries where only a single crop is grown per year.

During the coming generation every populous low-income country, and particularly India, China, Pakistan, Nigeria, and Indonesia, the five nations that will account for more than 40% of the world's population increase by the year 2020 [*United Nations Organization* (*UNO*), 1998], will have to increase its N fertilizer applications. In contrast, European applications have been declining, and they will remain stable or will decline further with the removal of excessive farming subsidies and with limits imposed due to environmental concerns (see *Jongbloed and Henkens* [1996] and European Fertilizer Manufacturers Association's Code of Best Agriculture Practice-Nitrogen available as http://www.efma.org; referred to as EFMA data).

3.8. Fertilizer Recovery Rates

Calculated inputs sum up to 151-186 Tg N, with additions close to 170 Tg N yr^{-1} being most likely (Table 5). Dividing the element removed in crops by the estimates of total N inputs results in average global recovery rates of 46-56% (mean 50%). These rates serve as an important means of checking the validity of presented calculations. Data from numerous N intake studies (most reliably from those using ^{15}N) constrain the range of plausible recovery rates (their global mean could not be below 35% or above 65%), and they also help to delimit the most likely rate. Analyses of 30 (mostly temperate zone) agroecosystems showed that 2/3 of them had N recovery rates below 50%, while the most efficient cropping absorbed nearly 70% of applied N [*Frissel and Kolenbrander,* 1978]. Where fertilization rates were below 150 kg N ha^{-1}, uptake efficiencies were as high as 60-65% but with higher N application recovery rates scattered around 50%.

An extensive European survey found the following N uptake efficiencies: high-yielding wheats in France 39-57% (from urea) and 38-70% (from ammonia), in England 52-65%, and in the Netherlands 52-62%, but in Portugal just 27-40%, and in Greece only 18-37% [*Jenkinson and Smith,* 1989]. In Asian rice, typically recovery rates are between 30-35%; they may be as low as 20% and only rarely do they exceed 40% of the applied nutrient [*De Datta,* 1995; *Cassman et al.,* 1996]. North American corn recovers between 40-60% [*Reddy and Reddy,* 1993], and leguminous crops, particularly forages, incorporate generally more of the available (and largely self-supplied) N, usually anywhere between 50-90% [*Peoples and Herridge,* 1990]. Recovery rates of N added to soils by atmospheric deposition and irrigation water are similar to those of inorganic N.

Multiplying conservative means of N recovery rates for major crop categories, 35% for rice and for rainfed crops grown in drier climates, 55% for crops in humid climates, 65% for legumes, and 75% for cropland forages, by the FAOSTAT Data areas sown to these crops results in a weighted recovery average of about 50%, an excellent confirmation of the calculated mean. While we may conclude with a high degree of confidence that 1/2 of all N added annually to the world's croplands is not incorporated in harvested biomass, we have much less confidence about apportioning this huge N loss among the major processes that remove N from agroecosystems to the atmosphere and to both underground and surface waters.

Table 5. Annual N Inputs to the World's Croplands During the Mid-1990s

Inputs	Minimum	Mean	Maximum
Seeds	2	2	2
Atmospheric deposition	18	20	22
Irrigation water	3	4	5
Crop residues	12	14	16
Animal manures	16	18	20
Biofixation	25	33	41
Inorganic fertilizers	75	78	80
Total	151	169	186

All values are in Tg N yr^{-1}. Derivations of individual input rates is described in the text. All figures are rounded to avoid the appearance of unwarranted precision.

4. Nitrogen Losses

Both nitrification and denitrification remove soil N as NO and N_2O, and denitrification, the closing arm of the biospheric N cycle, restores N_2 to the atmosphere. Volatilization of NH_3 is responsible for large N losses from both animal manures and from all ammonia fertilizers. Leaching of highly soluble nitrates and soil erosion transfers often large amounts of N to ground water, and to streams, lakes and coastal waters where the nutrient can cause serious eutrophication; erosion can also remove a great deal of organic N. There are also various, and often considerable, N losses from tops of plants before the harvest.

4.1. NO_x and N_2O Emissions

Two key flows in the biospheric nitrogen cycle, bacterial nitrification (oxidation of NH_4^+) and denitrification (reduction of NO_3^- or NO_2^-), are the main sources of NO_x (mostly NO) and N_2O emitted from soils. Chemical denitrification and other kinds of bacterial metabolism involving oxidation or reduction of N also yield trace amounts of the two gases. Nitrification rates are regulated primarily by the availability of NH_4^+; denitrification is controlled mainly by temperature, precipitation, and soil texture, C availability and pH [*Bouwman et al.,* 1993].

Rates of fertilization, crop varieties, and tillage practices are the most important management factors determining the emissions. Water-filled pore space (WFPS) appears to be the key determinant of the relative fluxes of the two gases: NO fluxes peak with 30-60% WFPS, while the N_2O flows peak when WFPS is between 80-90% [*Veldkamp et al.,* 1998]. Rates of NO and N_2O emissions from agricultural soils are so highly variable that annual fluxes (derived mostly from short-term measurements) range over several orders of magnitude [*Williams et al.,* 1992; *Bouwman,* 1996].

Shepherd et al. [1991] measured NO-N emissions equal to 11% of added N in a fertilized soil in Ontario. Typical NO-N losses are considerably lower, ranging between 0.02-5.7% of the nutrient applied in fertilizers [*Harrison et al.,* 1995; *Veldkamp et al.,* 1998]. Consequently, annual means of NO emissions from agricultural soils derived from long-term measurements range from mere traces to nearly 10 kg N ha^{-1}. *Potter et al.* [1996] used the average rates of 1.5 kg N/ha as NO in global simulations; *Davidson and Kingerlee* [1997] found the mean of 3.6 kg N ha^{-1} for temperate fields worldwide; *Davidson et al.* [1998] used the mean of 8.8 kg N ha^{-1} for the soils of the southeastern United States but admitted it may be somewhat high. Given this great uncertainty I will assume a conservative range of 0.5-4 kg N ha^{-1}, resulting in annual global releases of 1-6 Tg NO-N from the world's cultivated soils.

Recorded N_2O emissions also range over several orders of magnitude. Annual means for emissions from soils that have not been recently fertilized range mostly between 1-5 kg N ha^{-1}, and the average amount of fertilizer-induced N_2O emissions increases over time, and it ranges mostly between 0.5-2% of initial N applications [*Harrison et al.*, 1995]. *Veldkamp et al.* [1998] recorded rates as high as 6.8% in tropical pastures on soils developed from volcanic ash, and *Shepherd et al.* [1991] found conversions equal to 5% of applied fertilizer. Manure applications can increase N_2O emissions, from both nitrification and denitrification, several-fold compared to fluxes from soil receiving some, or no, inorganic fertilizer [*Li et al.*, 1994].

In their model of global N trace gas emissions, *Potter et al.* [1996] used the rate of 0.8 kg N/ha as N_2O as an average flux from cultivated soils; this would produce annually about 1.2 Tg N. *Bouwman* [1996] recommended the mean of 1.25% of N fertilizer applications, in addition to 1 kg N_2O-N/ha of background emissions, to be used for large-scale, order-of-magnitude quantifications; this would translate to about 2.5 Tg N of annual emissions.

Li et al. [1996], using their detailed denitrification-decomposition model, estimated total N_2O emissions from the U.S. croplands at 0.5-0.74 Tg N yr^{-1} or 3-4.6 kg N ha^{-1}; extending the highest rate to the global cropland would result in emissions of up to 7 Tg N yr^{-1}. I will use the range of 1-7 Tg N for global N_2O-N emissions from agricultural soils. Given the enormous variability of NO and N_2O fluxes from agricultural soils and from applied fertilizers, the range emerging from these estimates, 2-13 Tg yr^{-1}, is not excessive.

4.2. Complete Denitrification

Estimating annual return of N_2 produced by complete denitrification is by far the most elusive task in quantifying principal fluxes of the global N cycle. Obviously, the process is governed by the same factors as N_2O generation, with rates highly dependent on soil water content (high WFPS) and available soil carbon. High concentrations of soil NO_3^- inhibit the final conversion step from N_2O to N and lower the $N_2:N_2O$ ratios [*Weier et al.*, 1993]. Reliable direct measurements of complete denitrification in farmlands are rare, and denitrification fluxes in field or regional N balances are commonly estimated as residual values after accounting for other major N losses.

As with NO and N_2O emissions, the enormous variability of fluxes from different soils and under different environmental conditions makes the choice of any typical or average value highly questionable. However, with N_2 emissions being so much larger than the combined NO and N_2O flux, there is much greater opportunity for underestimating or exaggerating the overall flux by relatively small shifts in assumptions. *Galloway et al.* [1995] based their estimates of total global denitrification on $N_2:N_2O$ ratios between 14-32, but, as explained in the following two paragraphs, the most likely ratios for agricultural soils may be much lower.

After 5 days of measurements in four kinds of benchmark soils, *Weier et al.* [1993] found $N_2:N_2O$ ratios ranging from as low as 0.2 to as much as 245 and, not surprisingly, concluded that using an average ratio for estimating total denitrification from N_2O field measurements cannot be recommended. Still, their information, combined with other data, can be used to narrow down the uncertainty. Most of their ratios fit between 1-20; simple mean is about 8; and weighted mean (after discarding a handful of the most extreme outliers) is about 5. Taking the last value as the minimum ratio would, with annual emissions of at least 2 Tg N_2O-N, generate no less than 10 Tg N or about 10% of all N particularly susceptible to relatively rapid denitrification (that is N introduced in atmospheric deposition, irrigation water, animal manures, and inorganic fertilizers).

Similar rates have been reported in relatively rare studies of total denitrification. *Rolston et al.* [1978] found total denitrification fluxes equal to 11-14% of applied N (inorganic and manure) on cropped sites, and *Ryden et al.* [1979] measured an average loss of about 15% from a heavily fertilized and irrigated farmland, with average $N_2:N_2O$ ratios between 5.6-7.4. *Svensson et al.* [1991] found seasonal losses between 2-7% just for the applied inorganic N. Losses equal to 10-15% of N most susceptible to denitrification would produce annual fluxes between 11-18 Tg N, and, uncertain as this estimate is, it is unlikely that the actual flux would be lower than the mean of this range. Obviously, much more N_2 is eventually returned from agroecosystems to the atmosphere by denitrification of N compounds which were removed from fields by leaching and soil erosion, and carried away in harvested feed and food crops.

4.3. Volatilization of NH_3

Relatively large amounts of NH_3-N that can be lost during days and weeks following applications of ammonia fertilizers and animal manures have been a long-standing concern among agronomists [*Terman*, 1979; *Jayaweera and Mikkelsen*, 1991]. Reported shares of initial N loss are as high as 46% within a week for urea and 80% for animal urine in just 3 days [*Hargrove*, 1988]. Dry, calcareous soils, surface applications of shallow incorporation of fertilizers, and manures and high temperatures promote the loss in rain-fed fields.

Volatilization losses are particularly high when ammonia fertilizers are broadcast directly onto flooded soils [*Jayaweera and Mikkelsen*, 1991]. Principal factors promoting the process are shallow waters, their alkaline pH, high temperature and high NH_4^+-N concentration, and higher wind speeds. Consequently, the highest volatilization losses are in heavily fertilized paddy fields in the tropics; dominance of urea in rice farming makes matters worse as the *p*H of noncalcareous soils is temporarily elevated after its application [*Hargrove*, 1988].

Volatilization losses of the order of 10% within 1-3 weeks of N applications are common, and recorded maxima are above 60% or even 70%. Volatilization from non-NH_3 fertilizers is minimal, and large-scale (national or global) means used to calculate annual losses have been as low as 1% [*ApSimon et al.*, 1987] for the UK and as high as 11% [*Bouwman et al.*, 1997] for the world. Given the still increasing use of urea and higher intensity of N applications in humid and warm environments in general and in Asia's monsoonal rice fields in particular, worldwide losses of 8-12% (10%) of N applied in inorganic fertilizers would produce annual fluxes of 6-10 (8) Tg NH_3-N yr^{-1}. To these losses must be added volatilization of animal wastes applied to cropland.

Numerous experimental and fields studies show losses of at least 10-15% of the initial N content within days or weeks after application of animal manures, and with surface applications of fresh cattle or pig slurries the losses may be as much as 60-80%

of NH_3-N or some 30-40% of total N [*Terman*, 1979; *Hansen and Henriksen,* 1989]. This means that in addition to the conservatively estimated preapplication losses, there will be further volatilization losses of at least 15-20% of N applied in animal manures or about 2-4 Tg N. This would bring the total loss of the nutrient applied in inorganic and organic fertilizers to at least 11 (8-14) Tg N or roughly between 5-10 kg N ha^{-1}.

Net effects of annual deposition-volatilization fluxes of NH_3 over croplands are highly variable. Changes both in direction and magnitude take place on daily as well as seasonal basis, and in spite of often substantial NH_3 deposition some sites may actually show small (<1 kg N ha^{-1} yr^{-1}) net NH_3 emissions [*Sutton et al.,* 1993]. More importantly, there is a sizeable NH_3 flux from tops of maturing plants, and I will account for these emissions at the end of this survey of N losses.

4.4. Leaching

Leaching rates depend on levels of fertilization, compounds used (NH_3 leaches very little in comparison to readily soluble NO_3), soil thickness and permeability, temperature, and precipitation. The single most important land management determinant of the intensity of leaching is the presence and quality of the ground cover [*Hill*, 1991]. Everything else being equal, leaching from bare, fallow, or freshly ploughed soils is much higher than beneath row crops, which, in turn, is considerably higher than from soils under such dense cover crops as legume-grass mixtures whose roots can take up large amounts of added N.

Relatively abundant information on nitrate exports in streams and groundwaters [*Cole et al.,* 1993; *Canter*, 1997] is of little use in assessing N leaching from agricultural land as there is no reliable way to separate the contributions of atmospheric deposition, sewage and industrial processes from the flux originating in inorganic fertilizers and animal manures. Because of its effects on the quality of surface and ground waters, nitrate leaching has received a great deal of research attention [*Addiscott et al.,* 1991; *Hill*, 1991; *Burt et al.,* 1993], but, as with many other N fluxes, leaching rates have been measured mostly over short periods of time (days to months after fertilizer application) and prorating these figures may underestimate long-term throughputs [*Valiela et al.*, 1997].

Annual leaching losses range from negligible amounts in arid and semi-arid fields to 20-30 kg N ha^{-1} in rainy temperate regions. Maxima of over 50 kg N ha^{-1} are not unusual in the most heavily fertilized crop fields of northwestern Europe and the U.S. Midwest. Even higher leaching losses have been reported in some irrigated crops [*Diez et al.,* 1997; *Prunty and Greenland.*, 1997], but losses in many Asian paddy fields are very low. A survey of 40 agroecosystems on three continents indicated that with applications of less than 150 kg N ha^{-1} leaching equaled about 10% of fertilizer N, while with additions of more than 150 kg N ha^{-1} about 20% of added N was lost [*Frissel and Kolenbrander*, 1978]. Losses of 15-25% of initial N were also measured with repeated applications of cattle feedlot manure [*Chang and Entz,* 1996]. Highest leaching rates, on sandy soils, may remove over 60% of N applied in manure [*Hansen and Djurhuus,* 1996].

I prepared two kinds of global estimates. The first one, assigning specific average leaching shares (5-25%) to inorganic N applications disaggregated by 11 major agricultural regions, ended up with a flux of about 13 Tg N yr^{-1}; the second one, applying typical annual leaching rates (5-30 kg N ha^{-1}) to the same regional set, came up with about 17 Tg N yr^{-1}. To these totals must be added the leaching losses from animal manures: with no less than 10-20% of applied N they would come to 1-3 Tg N yr^{-1}. Annual global leaching loss from agricultural land would be then about 17 (14-20) Tg N, averaging between 10-15 kg N ha^{-1}.

4.5. Soil Erosion

Recent concerns about soil erosion and its effects on crop productivity [*Pimentel*, 1993; *Pimentel et al.,* 1995; *Agassi*, 1995] have not been matched by reliable information about the actual extent and intensity of the process or, more accurately, about soil losses in excess of natural denudation. Global assessment of soil degradation [*Oldeman et al.*, 1990] estimated that about 750 Mha of continental surfaces are moderately to excessively affected by water erosion and 280 Mha by wind erosion, with deforestation and overgrazing being major causes; mismanagement of arable land was estimated to be responsible for excessive erosion on some 180 Mha of cropland, but no global quantification of excessive erosion was attempted.

Because erosion rates vary widely even within a single field, any large-scale generalizations are merely order-of-magnitude indicators. They also ignore the complexity of the process: much of the eroded soil is not lost to food production. *Larson et al.* [1983] found that very little or no eroded sediment leaves the cultivated land in areas with gentle relief without any major surface outlet, the landscape common in the north-central United States; and when the sediment leaves the land, most of it may be deposited downstream as colluvium or alluvium [*Trimble*, 1975]. Consequently, even if croplands were the only sources of eroded soil, the amount of N exported from the fields could not be reliably quantified by using the relatively abundant data on the transport of suspended sediment to the ocean.

Short-term rates of cropland erosion have been measured and estimated with varying degrees of accuracy for many locales around the world. Their extremes range from negligible losses in rice paddies to more than 200 metric tons (t) ha^{-1} on steep tropical slopes and in the world's most erodible soils of China's Loess Plateau [*Liu*, 1988; *Hamilton and Luk*, 1993]. Average soil erosion estimates for larger areas are rarely available. Nationwide U.S. inventories, begun in 1977, are a major exception: Recent annual means were just above 15 t ha^{-1}, with about 60% contributed by water erosion, and the preliminary value for 1997 is about 14 t ha^{-1} [*Lee*, 1990] (see also Cropland acreage, soil erosion, and installation of conservation buffer strips: Preliminary estimates of the 1997 National Resources Inventory available as http://www.nhg.nres.usda.gov/land). The first nationwide approximation for India indicates losses of at least 13 t ha^{-1} for water erosion alone [*Singh et al.,* 1992], and regional estimates for China [*Zhu*, 1997] imply a national mean of about 30 t ha^{-1}.

Global mean of 20 t ha^{-1} implies annual removal of about 30 Pg of soil, a conservative total given the high erosion rates in parts of Africa, Latin America, and Asia [*Oldeman et al.*, 1990; *Pimentel*, 1993]. A disaggregated calculation using specific rates for North America (17 t ha^{-1}), Europe (15 t ha^{-1}), Asia (35 t ha^{-1} except for rice paddies where erosion is generally negligible), Africa, and Latin America (30 t ha^{-1}) ended up with about 35 Pg. With at least a quarter of this soil redeposited on adjacent cropland or on more distant alluvia, the loss would be 22-25 Pg yr^{-1}. Soil N content is highly variable even within a single field, ranging over an order of magnitude (1.5-17 t N ha^{-1}) in cropped soils [*Stevenson*, 1986]. A value just short of 0.1% N is close to the current Chinese mean [*Lindert and Wu*, 1996], while the U.S. average is around 0.125% [*Sprent*, 1987]. A conservative range of 0.08-0.1% N would mean that 22-25 Pg of eroded soil would carry away about 20 (18-25) Tg N yr^{-1}.

4.6. Losses From Tops of Plants

Although rarely discussed, these losses add up to one of the largest global N fluxes. They occur mostly within 2 or 3 weeks after anthesis (full bloom), and at harvest time the crops may commonly have between 15 and 30% less N than was their peak content. Neither translocation to roots nor root exudates are good explanations. A major part of the losses must be due to the shedding of various plant parts (pollen, flowers, and leaves), leaching of N from senescing leaves, and to heterotrophic (microorganismic, insect and bird) grazing. All of these losses (except for windborne pollen) can be seen as merely internal redistributions of N, as the litter fall, leaching and herbivory will return the nutrient to soils, and do not have to be quantified. Although both reduced and oxidized N compounds are emitted by plant tops, most of the N loss from tops of plants is due to volatilization of NH_3 [*Francis et al.*, 1993].

As with most other N losses, there is a considerable variability: Measured postanthesis losses in wheat ranged from less than 6 to 80 kg N ha^{-1} or from less than 8% to almost 60% of N present at anthesis [*Daigger et al.*, 1976; *Harper et al*, 1987; *Kanampiu et al.*, 1997]. Postanthesis declines ranging mostly between 20-50 kg N ha^{-1} were measured in other cereal crops (rice, sorghum, and barley), with daily losses as high as 2 kg N ha^{-1} [*Wetselaar and Farquhar*, 1980], and *Francis et al.* [1993] found rates as high as 45-81 kg N ha^{-1} from corn, equal to 52-73% of all N unaccounted for by standard balance calculations. Even if the typical loss would be just 10 kg N ha^{-1}, the world's croplands would lose annually some 15 Tg N, a flux similar to NO_3-N losses due to leaching.

At 20 kg N ha^{-1}, the global loss of around 30 Tg N yr^{-1} would make the N loss from tops of plants about as large a route of N egress from croplands as denitrification. Given the magnitude of this flux, it is imperative that all N balance studies should consider this neglected variable before attributing any unaccounted losses to unknown factors or to higher rates of denitrification or leaching [*Kanampiu et al.*, 1997]. Because we do not know to what extent the gaseous N losses from tops of plants may have been captured by previously outlined estimates of NH_3 and NO_x fluxes, I will assume that no more than 10 (5-15) Tg N yr^{-1} are lost in this way.

5. Nitrogen Balances

Tabular recapitulation of the estimated inputs and losses (Table 6) shows that global croplands receive 169 (151-186) Tg N yr^{-1} and that between 143-190 (mean of about 165) Tg N yr^{-1} are removed in harvested crops and due to a variety of N losses. Figure 1 shows the fluxes involved in global crop production within a general N cycle model centered on plants. The fact that the means of my conservatively estimated inputs and outputs (169 and 165 Tg N) are merely 2% apart is not a strong indication that global agroecosystem is in N equilibrium.

Indeed, the very notion of any direct, immediate balancing of N inputs and removals on an annual basis is misleading as most of the added N is not used directly by crops (or weeds): The bulk of the nutrient recovered by plants becomes available only after extensive turnover through the two opposing processes of immobilization and mineralization. Experiments using ^{15}N-labeled fertilizers are the best way to reveal this complexity: At the growing season's end, a large share (commonly up to 40%, sometimes more than 50%) of the fertilizer N is immobilized in the soil's biomass, and the amount of (previously immobilized and then mineralized) soil N utilized by crops can be 3-6 times as high as the nutrient drawn directly from the applied fertilizer [*Stevenson*, 1986; *Reddy and Reddy*, 1993].

High C/N ratios of most crop residues, commonly above 50 and as high as 150, are particularly conducive to rapid N immobilization. Some residual N will be bound in persistent humus compounds and be unavailable to plants for decades or centuries. How fast the short-lived fraction will cycle depends on the activity of microbial decomposers (which is highly temperature- and moisture-dependent) and on the availability of other sources of N.

5.1. Gains or Losses?

Are then the world's croplands gaining or losing nitrogen? Comparisons of my minimum input and output estimates show an appreciable net gain (8 Tg N); those of the two maximum values show a small net loss (4 Tg N); cross-comparisons of minimum and maximum estimates indicate relatively large gains of 43 Tg N yr^{-1} (about 28 kg N ha^{-1}) or an almost as large loss of 39 Tg N yr^{-1} (about 25 kg N ha^{-1}). The two extreme comparisons are not plausible: We have no indication that the world's agricultural soils would be experiencing a net gain or loss as high as 40 Tg N yr^{-1}. In contrast, either of the first two possibilities is entirely plausible, as they imply annual increments or removals of only a few kg N ha^{-1}. Although we cannot exclude the possibility of a small but chronic N loss caused by improper agronomic practices and by accelerated natural soil degradation in many farming regions, the likelihood of small gains appears to be much higher.

An international comparison of N balances in almost 40 agroecosystems on three continents found gains in soil organic N

Table 6. Annual Balances of N Flows in the World's Croplands During the Mid-1990s

Flows	Minimum	Mean	Maximum
Inputs	151	169	186
Outputs	143	165	190
Harvested plants	85	85	85
Losses			
NO emissions	1	4	6
N_2O emissions	1	4	7
N_2 emissions	11	14	18
NH_3 volatilization	8	11	14
NO_3^- leaching	14	17	20
Soil erosion	18	20	25
Losses from tops of plants	5	10	15
Balance	+8	+4	-4

All values are in Tg N yr^{-1}. Inputs are from Table 5; derivations of individual output rates are described in the text. All figures are rounded to avoid the appearance of unwarranted precision.

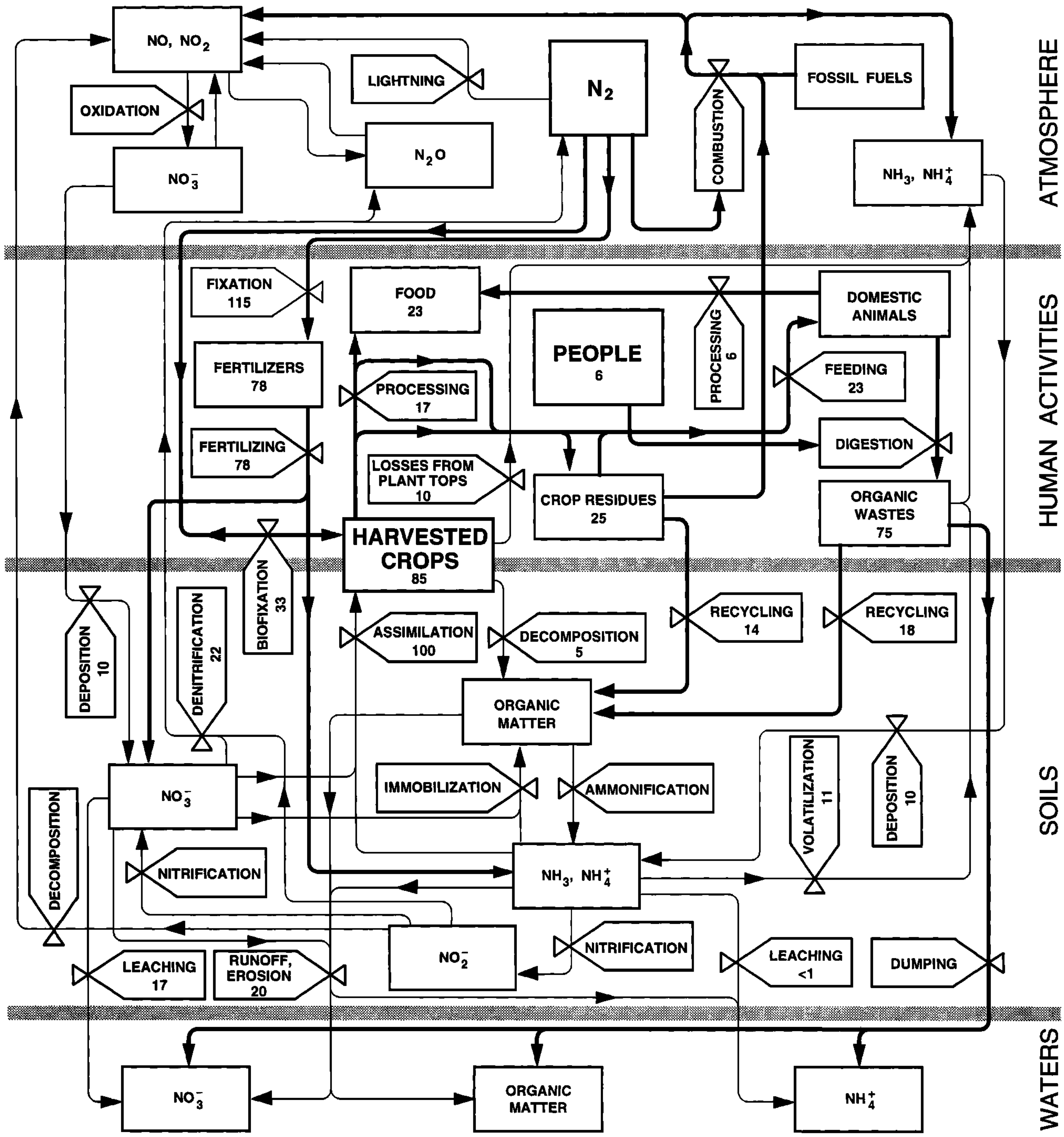

Figure 1. Simplified graph of the biospheric nitrogen cycle centered on agricultural crops. Only the storages (rectangles) and flows (valves) discussed in this paper are quantified (all values are in Tg N yr^{-1}). Thick lines identify fluxes directly affected by human actions.

(the pool usually containing > 90% of soil N) in almost 60% of the cases, with a mean annual increase of 35 kg N ha^{-1} and no change in about 1/3 of the balances [*Floate*, 1978]. In Europe, two recent national agricultural N balances, for Germany and the Netherlands, found average annual accumulations of 47 and 38 kg N ha^{-1}, respectively [*van Dijk*, 1993]. *Zhu* [1997] has shown that nitrogen balance in China's agriculture has been positive between the early 1950s and the late 1980s. Given the rapid rate of increase in the country's average N applications, there is no doubt that these gains have continued during the 1990s: My preliminary estimates put them now at about 25 kg N ha^{-1} yr^{-1}. Organic farming can produce the same results: Comparison of long-term balance sheets for the Rothamsted Continuous Wheat Experiment show that the plot receiving 35 t ha^{-1} of farmyard manure more than doubled its total N soil content in 115 years, gaining the nutrient at an annual rate of about 33 kg N ha^{-1}, and even the plot that did not receive any N fertilizers had no long-term N loss [*Jenkinson*, 1982].

Such gains are not surprising given the often high rates of N applications and the well-known sequestration of a significant share of the added nutrient in long-lived fractions of soil organic matter. As many properly farmed soils around the world are gaining annually 25-35 kg N ha^{-1} (that is of the order of 0.25-0.5% of total soil N stores), these increments could outweigh undoubted N losses from deteriorating soils, particularly in Africa where nutrient mining is most widespread or in highly erodible soils of China's Loess Plateau. Consequently, it is quite likely that the net result is a nonnegligible global N gain of 4-8 Tg N yr^{-1}. While this is welcome as far as the maintenance of the soil quality is concerned, positive N balances carry risks of increased N losses.

Tracing major N inputs to the world's agroecosystems and subsequent N losses associated with food production makes it easier to answer a key question posed by *Galloway et al.* [1995]: Where is N fixed by humans going? It also makes it possible to suggest the most effective ways of reducing the inputs of anthropogenic N into the biosphere.

5.2. Nitrogen Flows in Food Production

My calculations show that almost half of all N received annually by the world's croplands (46%, range 43-50%) comes from synthetic fertilizers. This high dependence is irreversible, and it is bound to increase even if the global population, as some forecasts suggest [*UNO*, 1998], were to stabilize at a relatively low level (less than 10 billion people) during the next two to three generations. While there are many energy sources that can replace fossil fuels, whose combustion is the main cause of human alteration of carbon cycle, there appears to be no imminent alternatives to our high, and increasing, reliance on the nitrogen fixed by the Haber-Bosch process [*Smil*, 1991].

Direct additions of all newly fixed anthropogenic N, inorganic fertilizers (75-80 Tg N) and symbiotic biofixation by food, feed, and forage legumes (22-32 Tg N), added up to 105 (97-112) Tg N yr^{-1} during the mid-1990s. As recycling, in seeds, crop residues, and animal manures, and irrigation water returned 38 (33-43) Tg N yr^{-1}, human management was responsible for inputs totaling 143 (130-155) Tg N yr^{-1} and equal to about 85% of all N received by the world's agricultural land. Natural atmospheric deposition and no more than 1/3 of biofixation (by free-living bacteria and cyanobacteria) are the only inputs not directly manipulated by humans. If the contributions from atmospheric deposition attributable to NO_x and NH_3 from combustion and agriculture are included, then more than 90% of all N received by the world's farmlands originate in human activities.

With 50% (46-56%) of all N inputs assimilated by harvested crops, we have to account for 50% (44-54%) of added N. Direct losses to the atmosphere (as N_2, NO_y, and NH_x) add up to 26-60 Tg N, while 32-45 Tg N, or about 20-25% of all initial inputs, enter ground and surface waters. About 50 Tg N yr^{-1} were removed from fields in harvested crops. Subtracting N in seeds (2 Tg N), postharvest losses (5 Tg N), nonfood uses (1 Tg N), and animal feed (23 Tg N) leaves 17 Tg N (after 2 Tg N of retail and household losses) in plant foods. Adding 6 Tg N in animal foods (whose N comes not only from concentrate feeds and cropland pastures but also from permanent grasslands and, about 1 Tg, from aquatic catches) makes up 23 Tg N yr^{-1} in the global food consumption.

Assuming an average body mass of 45 kg (weighted mean taking into account the age-sex structure of the world's population) and using typical lipid, bone and protein shares in human bodies [*Bailey*, 1982] results in about 6 Tg N stored in bodies of the world's 6 billion people. With the global population growing annually by almost 80 million people, the net increase of this pool is less than 80 Gg N yr^{-1}, or merely 0.3% of 23 Tg N consumed in food. More than 99% of ingested N is thus excreted, and advancing urbanization, as well as higher concentration of meat, egg, and milk production, means that increasing shares of this waste are released, directly or indirectly, into waters. Subsequent fate of this aqueous N ranges from benign (relatively rapid denitrification) to worrisome (nitrate contamination of aquifers, coastal eutrophication).

Another remarkable feature of anthropogenic N flows is the extent to which the nutrient is now moved among countries and continents by fertilizer and agricultural trade. During the mid-1990s, about 22 Tg N in inorganic fertilizers (more than a quarter of the applied total) were sold annually on the international market, and the traded farm products contained nearly 20% (almost 10 Tg N yr^{-1}) of all N incorporated in food and feed crops, mostly in cereals, oilseed cakes, and soybeans (FAOSTAT). The share of traded N will also increase in the future, as will the fraction of all N traded intercontinentally.

5.3. What Can be Done?

The best way to manage the continuing human interference in N cycle is to maximize the efficiencies of N used in crop production. Fortunately, there are many realistic possibilities for lowering the input of anthropogenic N in agricultural production and for reducing its field and postharvest losses [*Munson and Runge,* 1990; *Fragoso and van Beusichem,* 1993; *Dudal and Roy,* 1995; *van der Voet et al.,* 1996; *Prasad and Power,* 1997]. Soil testing, choice of appropriate fertilizing compounds, maintenance of proper nutrient ratios, and attention to timing and placement of fertilizers are the most important direct measures of universal applicability. Indirect approaches which can either reduce the need for synthetic fertilizers or increase the efficiency of their use rely primarily on greater contributions by biofixation accomplished by more frequent planting of leguminous crops or by optimizing conditions for other diazotrophs.

Improving a variety of agronomic practices, including those not directly connected with fertilization, can further reduce N losses. Overall N uptake can be improved by planting N-responsive varieties, maximizing recycling of organic wastes and integrated use of organic and synthetic fertilizers, by minimizing soil erosion, by assuring adequate moisture supply, and by controlling pests. The value of this integrated approach has been recognized by Codes of Best Agricultural Practice mandated by the European Union's Directive of December 1991 and designed to protect fresh, coastal, and marine waters against nitrate pollution from diffuse sources.

These codes recommend that applications do not exceed crop needs (after the contributions from organic sources are taken properly into account); these codes ask that soils should not be let bare during rainy periods and that nitrates present in soil between crops should be limited through planting of trap crops [see *Ignazi*, 1996; and EFMA data].

Effects of these improvements can be impressive. *El-Fouly and Fawzi* [1996] concluded that proper N:P:K ratios based on soil testing and plant analysis and adjusted to the prevailing cropping sequence could raise typical Egyptian yields by 20% without using more nitrogen. Gains in fertilizer efficiency could already be seen in the U.S. record since the early 1980s: Average yields of principal field crops have continued to increase at a faster rate than the total applications of fertilizer N [*Munson and Runge*, 1990; *USDA*, 1998]. Good agronomic practices should raise the average N use efficiency by at least 25-30% during the next two generations, i.e., to average uptakes of at least 50-55% in low-income countries and to around 65-70% in affluent nations.

Even if the utilization of nitrogen from other sources would remain constant, such higher fertilizer efficiencies would use 10-12 Tg of the currently wasted N applications. In reality, an effective supply of nitrogen from organic wastes, biofixation, mineralization, and atmospheric deposition should also increase because of reduced N losses in soil erosion and because of more frequent rotations and more vigorous recycling. Again, relatively modest improvements would translate into impressive total gains: Reducing erosional losses by 20% would save roughly 4 Tg N from nonfertilizer sources, and expanding biofixation, and waste recycling by 10% would add another 5 Mt N. Cumulative effects of adopting well-proven and mostly low-cost measures aimed at increasing efficiency of nutrient uptake would be then equal to expanding effective N supply to crops by 20-22 Tg yr^{-1}.

These gains would make it possible to satisfy virtually all anticipated N demand needed to produce improving food supply for the world's population during the next generation without increasing the input of anthropogenic N in crop production. Such a stabilization would be the first step toward easing the human impact on the biospheric N cycle.

6. Conclusions

Of all the N flows involved in the world's crop production, only the inputs of the nutrient in inorganic fertilizers and N uptake by crops and their residues can be quantified with a high degree of accuracy. For this reason I presented all N flows in the global agroecosystem as ranges rather than as single values. However, these extreme values do not represent the total uncertainty in the estimates: They merely indicate a most likely range based on fairly conservative assumptions. This makes it unlikely that actual biospheric flows would be lower than the lowest estimates presented here. At the same time, it is likely that in a number of cases, the real fluxes may be higher than the highest given estimates; indeed several published rates are outside the ranges presented in Tables 5 and 6.

With these caveats in mind, it appears that the world's croplands receive about 170 Tg of fixed N yr^{-1}, with nearly 9/10 of this total coming from managed, anthropogenic inputs. Cropping thus represents by far the most important human interference in the biospheric N cycle, far ahead of the combustion of fossil fuels. Only about half of all fixed N reaching the fields is assimilated by crops and their residues; this conclusion is well supported by numerous studies tracing the recovery of N in cropping.

Losses of agricultural N thus amount to about half of all inputs, but their apportioning remains very uncertain. Complete denitrification, volatilization, leaching, and soil erosion are the principal routes of N removal, and the best available evidence indicates that contributions of these four processes are rather similar, with every one of them removing annually between 10-20 Tg N from the world's crop fields. Given the enormous spatial and temporal variability of these flows, it will not be easy to come up with reliable estimates on the global scale.

Fortunately, these large N losses introducing excessive amounts of reactive N into the biosphere, can be significantly reduced by better ways of fertilizing and by appropriate agronomic practices. While the world's dependence on the Haber-Bosch fixation of ammonia will have to increase during the first half of the 21st century, impacts on the global nitrogen cycle could be kept to tolerable levels.

References

Addiscott, T.M., A.P. Whitmore, and D.S. Powlson, *Farming, Fertilizers and the Nitrate Problem*, 170 pp., CAB Int., Wallingford, U.K., 1991.

Agassi, M. (Ed.), *Soil Erosion, Conservation, and Rehabilitation*, 424 pp., Marcel Dekker, New York, 1995.

Andreae, M.O., Biomass burning: Its history, use, and distribution and its impact on environmental quality and global climate, in *Global Biomass Burning: Atmospheric, Climatic, and Biospheric Implications*, edited by J.S. Levine, pp. 3-21, MIT Press, Cambridge, Mss., 1991.

ApSimon, H.M., M. Kruse, and J.N.B. Bell, Ammonia emissions and their role in acid deposition, *Atmos. Environ., 21*, 1939-1946, 1987.

Ayanaba, A., and P.J. Dart (Eds.), *Biological Nitrogen Fixation in Farming Systems of the Tropics*, 377 pp., John Wiley, New York, 1977.

Bailey, S.M., Absolute and relative sex differences in body composition, in *Sexual Dimorphism in Homo Sapiens*, edited by R.L. Hall, pp. 363-390, Praeger, Westport, Conn., 1982.

Barreveld, W.H., *Rural Use of Lignocellulosic Residues*, 89 pp., Food and Agric. Org., Rome, 1989.

Bath, D., J. Dunbar, J. King, S. Bery, and S. Olbrich S., Byproducts and unusual feedstuffs, *Feedstuffs, 69*(30), 32-38, 1997.

Berner, E.K., and R.A. Berner, *Global Environment*, 376 pp., Prentice-Hall, Englewood Cliffs, N.J., 1996.

Blair, G., D. Catchpoole, and P. Horne, Forage tree legumes: Their management and contribution to the nitrogen economy of wet and humid tropical environments, *Adv. Agron., 44*, 27-54, 1990.

Boddey, R.M., O.C. de Oliveira, S. Uequiaga, F.L. de Olivares, V.L.D. Bladani, and J. Dobereiner, Biological nitrogen fixation associated with sugar cane and rice: Contributions and prospects for improvement, *Plant Soil, 174*, 195-209, 1995.

Bouwman, A.F., Direct emissions of nitrous oxide from agricultural soils, *Nutrient Cycling Agroecosystems, 46*, 53-70, 1996.

Bouwman, A.F., I. Fung, E. Matthews, and J. John, Global analysis of the potential for N_2O production in natural soils, *Global Biogeochem. Cycles, 7*, 557-597, 1993.

Bouwman, A.F., D.S. Lee, W.A.H. Asman, F.J. Dentener, K.W. Van Der Hoek, and J.G.J. Olivier, A global high-resolution emission inventory for ammonia, *Global Biogeochem. Cycles, 11*, 561-587, 1997.

Broughton, W.J., and S. Puhler (Eds.), *Nitrogen Fixation*, 348 pp., Clarendon, Oxford, England, 1986.

Burt, T.P., A.L. Heathwaite, and S.T. Trudgill, *Nitrate: Processes, Patterns and Management*, 444 pp., John Wiley, New York, 1993.

Canter, L. W., *Nitrates in Groundwater*, 263 pp., CRC Press, Boca Raton, Fa, 1997.

Cassman, K.G., S.K. De Datta, S.T. Amarante, S.P. Liboon, M.I. Samson, and M.A. Dizon, Long-term comparison of the agronomic efficiency and residual benefits of organic and inorganic nitrogen sources for tropical lowland rice, *Exp. Agric. 32*, 927-944, 1996.

Chang, C., and T. Entz, Nitrate leaching losses under repeated cattle feedlot manure applications in southern Alberta, *J. Environ. Qual., 25*, 145-153, 1996.

Choudhary, M., L.D. Bailey, and C.A. Grant, Review of the use of swine manure in crop production: Effects on yield and composition and on soil and water quality, *Waste Manage. Res., 14*, 581-595, 1996.

Cole, J.J., B.L. Peierls, N.F. Caraco, and M.L. Pace, Nitrogen loading of rivers as a human-driven process, in *Humans as Components of Ecosystems*, edited by M.J. McDonnell and S.T.A. Pickett, pp. 141-157, Springer-Verlag, New York, 1993.

Daigger, L.A., D.H. Sander, and G.A. Peterson, Nitrogen content of winter wheat during growth and maturation, *Agron. J., 68*, 815-818, 1976.

Danso, S.K.A., G. Hardarson, and F. Zapata, Misconceptions and practical problems in the use of ^{15}N soil enrichment techniques for estimating N_2 fixation, *Plant Soil, 152*, 25-52, 1993.

Davidson, E.A., and W. Kingerlee, A global inventory of nitric oxide emissions from soils, *Nutrient Cycling Agroecos.*, 48, 37-50, 1997.

Davidson, E.A., C.S. Potter, P. Schlesinger, and S.A. Klooster, Model estimates of regional nitric oxide emissions from soils of the southeastern United States, *Ecol. Appl., 8*, 748-759, 1998.

De Datta, S.K., Nitrogen transformations in wetland rice ecosystems, *Fert. Res., 42*, 193-203, 1995.

Diez, J.A., R. Roman, R. Caballero, and A. Caballero, Nitrate leaching from soils under a maize-wheat-maize sequence, two irrigation schedules and three types of fertilisers, *Agric. Ecosyst. Environ., 65*, 189-199, 1997.

Donald, C.M., and J. Hamblin, The biological yield and harvest index of cereals as agronomic and plant breeding criteria, *Adv. Agron., 28*, 361-405, 1976.

Dovrat, A., *Irrigated Forage Production*, 257 pp., Elsevier, New York, 1993.

Dudal, R., and R.N. Roy (Eds.), *Integrated Plant Nutrition Systems*, 426 pp., Food and Agric. Org., Rome, 1995.

Eady, R.R., The enzymology of biological nitrogen fixation, *Sci. Prog., 78*, 1-17, 1995.

Eghball, B., J.F. Power, J.E. Gilley, and J.W. Doran, Nutrient, carbon, and mass loss during composting of beef cattle feedlot manure, *J. Environ. Qual., 26*, 189-193, 1997.

El-Fouly, M., and A.F.A. Fawzi, Higher and better yields with less environmental pollution in Egypt through balanced fertilizer use, *Fert. Res., 43*, 1-4, 1996.

Erisman, J. W., *Atmospheric Deposition in Relation to Acidification and Eutrophication*, 405 pp., Elsevier, New York, 1995.

Farnham, D.E., and J.R. George, Harvest management effects in dinitrogen fixation and nitrogen transfer in red clover-orchardgrass mixtures, *J. Prod. Agric., 7*, 360-364, 1994.

Floate, M.J.S., Changes in soil pools, in *Cycling of Mineral Nutrients in Agricultural Ecosystems.*, edited by M.J. Frissel, pp. 292-295, Elsevier, New York, 1978.

Food and Agriculture Organization of the United Nations (FAO), *Current World Fertilizer Situation and Outlook 1994/95-2000/2001*, Rome, 1997.

Fragoso, M.A., and M.L. van Beusichem (Eds.), *Optimization of Plant Nutrition*, 682 pp., Kluwer, Norwell, Mass., 1993.

Frame, J., and O. Newbould, Agronomy of white clover, *Adv. Agron., 40*, 1-88, 1986.

Francis, D.D., J.S. Schepers, and M.F. Vigil, Post-anthesis nitrogen loss from corn, *Agron. J., 85*, 659-663, 1993.

Frissel, M.J., and G.J. Kolenbrander, The nutrient balances: Summarizing graphs and tables, in *Cycling of Mineral Nutrients in Agricultural Ecosystems.*, edited by M.J. Frissel, pp. 273-292, Elsevier, New York, 1978.

Galloway, J.N., W.H. Schlesinger, H. Levy II, A. Michaels, and J.L. Schnoor, Nitrogen fixation: Anthropogenic enhancement-environmental response, *Global Biogeochem. Cycles, 9*, 235-252, 1995.

Gifford, R.M., and L.T. Evans, Photosynthesis, carbon partitioning, and yield, *Ann. Rev. Plant Physiol., 32*, 485-509, 1981.

Giller, K.E., and K.J. Wilson, *Nitrogen Fixation in Tropical Cropping Systems*, 200 pp., CAB Int., Wallingford, U.K., 1991.

Giller, K.E., J.F. McDonagh, and G. Cadisch, Can nitrogen fixation sustain agriculture in the tropics?, in *Soil Science and Sustainable Land Management in the Tropics*, edited by J.K. Syers and D.L. Rimmer, pp. 173-191, CAB Int., Wallingford, U.K., 1994.

Gutteridge, R.C., and H.N. Shelton, *Forage Tree Legumes in Tropical Agriculture*, 389 pp., CAB Int., Wallingford, U.K., 1994.

Hamilton, H., and S. Luk, Nitrogen transfers in a rapidly eroding agroecosystem: Loess Plateau, China, *J. Environ. Qual., 22*, 133-140, 1993.

Hansen, E.M., and J. Djurhuus, Nitrate leaching as affected by long-term N fertilization on a coarse sand, *Soil Use Manage., 12*, 199-204, 1996.

Hansen, J. A., and K. Henriksen (Eds.), *Nitrogen in Organic Wastes Applied to Soils*, 381 pp., Academic, San Diego, Calif., 1989.

Hardarson, G., Methods for enhancing symbiotic nitrogen fixation, *Plant Soil, 152*, 1-17, 1993.

Hardarson, G., and S.K.A. Danso, Methods for measuring biological nitrogen fixation in grain legumes, *Plant Soil, 152*, 19-23, 1993.

Hargrove, W.L., Soil, environmental, and management factors influencing ammonia volatilization under field conditions, in *Ammonia Volatilization from Urea Fertilizers*, edited by B.R. Bock and D.E. Kissel, pp. 17-36, National Fertilizer Development Center, Muscle Shoals, Ala., 1988.

Harper, L.A., R.R. Sharpe, G.W. Langdale, and J.E. Giddens, Nitrogen cycling in a wheat crop: Soil, plant and aerial nitrogen transport, *Agron. J., 79*, 965-973, 1987.

Harrison, R.M., S. Yamulki, K.W.T. Goulding, and C.P. Webster, Effect of fertilizer application on NO and N_2O fluxes from agricultural fields, *J. Geophys. Res., 100*, 25,923-25,931, 1995.

Hay, R.K.M., Harvest index: A review of its use in plant breeding and crop physiology, *Ann. Appl. Biol., 126*, 197-216, 1995.

Heichel, G.H., Legume nitrogen: Symbiotic fixation and recovery by subsequent crops, in *Energy in Plant Nutrition and Pest Control*, edited by Z. Helsel, pp. 62-80, Elsevier, New York, 1987.

Heichel, G.H., and K.I. Henjum, Dinitrogen fixation, nitrogen transfer, and productivity of forage legume-grass communities, *Crop Sci., 31*, 202-208, 1991.

Henson, R.A., Measurements of N_2 fixation by common bean in Central Brazil as affected by different reference crops, *Plant Soil, 152*, 53-58, 1993.

Hill, M. (Ed.), *Nitrates and Nitrites in Food and Water*, 196 pp., Ellis Horwood, Chichester, England, 1991.

Hoogerbrugge, I.D., and L.O. Fresco, *Homegarden Systems: Agricultural Characteristics and Challenges*, 23 pp., International Institute for Environment and Development (IIED), London, 1993.

Ignazi, I.C., Code of best agricultural practices, *Fert. Res., 43*, 241, 1996.

Intergovernmental Panel on Climate Change (IPCC) and Organization for Economic Cooperation and Development (OECD), *Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories*, Org. for Econ. Coop. and Dev., Paris.

Jaworski, N.A., R.W. Howarth, and L.J. Hetling, Atmospheric deposition of nitrogen oxides onto the landscape contributes to coastal eutrophication in the Northeast United States, *Environ. Sci. Technol., 31*, 1995-2004, 1997.

Jayaweera, G.R., and D.S. Mikkelsen, Assessment of ammonia volatilization from flooded soil systems, *Adv. Agron., 45*, 303-356, 1991.

Jenkins B.M., S.Q. Turn, and R.B. Williams, Atmospheric emissions from agricultural burning in California: Determination of burn fractions, distribution factors, and crop-specific contributions, *Agric. Ecosyst. Environ., 38*, 313-330, 1992.

Jenkinson, D.S., The nitrogen cycle in long-term field experiments, *Philos. Trans. R. Soc. Lond., Ser. B, 296*, 563-571, 1982.

Jenkinson, D.S., and K.A. Smith, *Nitrogen Efficiency in Agricultural Soils*, 450 pp., Elsever, New York, 1989.

Jongbloed, A.W., and C.H. Henkens, Environmental concerns of using animal manure - The Dutch case, in *Nutrient Management of Food Animals to Enhance and Protect Environment*, edited by E.T. Kornegay, pp. 315-332, Lewis Publ., Boca Raton, Fa, 1996.

Jordan, T.E., and D.E. Weller, Human contributions to terrestrial nitrogen flux, *BioScience, 46*, 655-664, 1996.

Kanampiu, F.K., W.R. Raun, and G.V. Johnson, Effect of nitrogen rate on plant nitrogen loss in winter wheat varieties, *J. Plant Nutr., 20*, 389-404, 1997.

Kinzig, A.P., and R.H. Socolow, Human impacts on the nitrogen cycle, *Phys. Today, 47*, 24-31, 1994.

Kossila, V.L., Global review of the potential use of crop residues as animal feed, in *Better Utilization of Crop Residues and By-products in Animal Feeding: Research Guidelines*, edited by T.R. Preston et al., pp. 1-13, Food and Agric. Org., Rome, 1985.

Kuhlbusch, T.A., J.M. Lobert, P.J. Crutzen, and P. Warneck, Molecular nitrogen emissions from denitrification during biomass combustion, *Nature, 351*, 135-137, 1991.

Larson, W.E., F.J. Dixon, and R.H. Dowdy, The threat of soil erosion to long-term crop production, *Science, 219*, 458-465, 1983.

LaRue, T.A., and T.G. Patterson, How much nitrogen do legumes fix? *Adv. Agron., 34*, 15-38, 1981.

Lee, L.K., The dynamics of declining soil erosion rates, *J. Soil Water Conserv.*, *45*, 622-624, 1990.

Li, C., S.E. Frolking, R.C. Harriss, and R.E. Terry, Modeling nitrous oxide emissions from agriculture: A Florida case study, *Chemosphere*, *28*, 1401-1415, 1994.

Li, C., V. Narayanan, and R.C. Harriss, Model estimates of nitrous oxide emissions from agricultural lands in the United States, *Global Biogeochem. Cycles*, *10*, 297-306, 1996.

Lindert, P.H., and Wu, W.L., Trends in the soil chemistry of North China since the 1930s, *J. Environ. Qual.*, *25*, 1168-1178, 1996.

Liu, T.S., *Loess in China*, 224 pp., Springer-Verlag, New York, 1988.

Meelu, O.P., Y. Singh, and B. Singh, *Green Manuring for Soil Productivity Improvement,* Food and Agric. Org., Rome, 1994.

Misra, R.V., and P.R. Hesse, *Comparative Analyses of Organic Manures*, 97 pp., Food and Agric. Org.,, Rome, 1983.

Mosier, A.R., et. al., Assessing and mitigating N_2O emissions from agricultural soils, *Clim. Change*, *40*, 7-38, 1998.

Munson, R.D. and C.F. Runge, *Improving Fertilizer and Chemical Efficiency Through "High Precision Farming"*, 96 pp., Cent. for Int. Food and Agric. Policy, Univ. of Minn., St. Paul, 1990.

National Intelligence Council, *China Agriculture Cultivated Land Area, Grain Projections, and Implications*, 24 pp., Nat. Intelligence Council, Washington, D.C., 1997.

National Research Council (NRC), *Atlas of Nutritional Data on United States and Canadian Feeds*, 772 pp., Nat. Acad. of Sci., Washington, D.C., 1972.

Nevison, C.D., G. Esser, and E.A. Holland, A global model of changing N_2O emissions from natural and perturbed soils, *Clim. Change*, *32*, 327-378, 1996.

Neyra, C.A., and J. Dobereiner, Nitrogen fixation in grasses, *Adv. Agron.*, *30*, 1-38, 1977.

Nordstedt, R.A., Selecting a waste management system, in *Large Dairy Herd Management,* edited by H.H. Van Horn, and C.J. Wilcox, pp. 630-639, Am. Dairy Sci. Assoc., Champaign, Ill., 1992.

Oldeman, L.R., et al., *World Map of the Status of Human-Induced Soil Degradation: An Explanatory Note*, United Nations Environment Programme (UNEP), Nairobi, 1990.

Peoples, M.B., and D.F. Herridge, Nitrogen fixation by legumes in tropical and subtropical agriculture, *Adv. Agron.*, *44*, 155-223, 1990.

Peoples, M.B., D.F. Herridge, and J.K. Ladha, Biological nitrogen fixation: An efficient source of nitrogen for sustainable agricultural production?, *Plant Soil*, *174*, 3-28, 1995.

Pimentel, D. (Ed.), *World Soil Erosion and Conservation*, 349 pp., Cambridge Univ., New York, 1993.

Pimentel, D., et. al., Environmental and economic costs of soil erosion and conservation benefits, *Science*, *267*, 1117-1122, 1995.

Postel, S.L., G.C. Daily, and P.R. Ehrlich, Human appropriation of renewable fresh water, *Science*, *271*, 785-788, 1996.

Potter, C.S., P.A. Matson, P.M. Vitousek, and E.A. Davidson, Process modeling of controls on nitrogen trace gas emissions from soils worldwide, *J. Geophys. Res.*, *101*, 1361-1377, 1996.

Powers, W.L., G.W. Walingford, and L.S. Murphy, Formulas for applying organic wastes to land, *J. Soil Water Conserv.*, *30*, 286-289, 1975.

Prasad, R., and J.F. Power, *Soil Fertility Management for Sustainable Agriculture,* 356 pp., CRC Press, Boca Raton, Fa., 1997.

Prihar, S.S, and B.A. Stewart, Sorghum harvest index in relation to plant size, environment and cultivar, *Agron. J.*, *83*, 603-608, 1991.

Prunty, L., and R. Greenland, Nitrate leaching using two potato-corn N-fertilizer plans on sandy soil, *Agric. Ecosyst. Environ.*, *65*, 1-13, 1997.

Reddy, G.B., and K.R. Reddy, Fate of N-15 enriched ammonium nitrate applied to corn, *Soil Sci. Soc. Am. J.*, *57*, 111-115, 1993.

Roberts, S.R., J.E. Hill, D.M. Brandon, B.C. Miller, S.C. Scarduci, C.M. Wick, and J.F. Williams, Biological yield and harvest index in rice: Nitrogen response of tall and semidwarf cultivars, *J. Prod. Agric.*, *6*, 585-588, 1993.

Rolston, D.E., D.L. Hoffman, and D.T. Toy, Field measurements of denitrification, I, Flux of N_2 and N_2O, *Soil Sci. Soc. Am. J.*, *41*, 863-869, 1978.

Rulkens, W.H., and P.J.W. ten Have, Central processing of pig manure in the Netherlands, *Water Sci. Technol.*, *30*, 157-165, 1994.

Ryden, J.C., L.J. Lund, J. Lesy, and D.D. Focht, Direct measurements of denitrification from soils, II, Development and application of field methods, *Soil Sci. Soc. Am. J.*, *43*, 110-118, 1979.

Shepherd, M.F., S. Barzetti, and D.R. Hastie, The production of atmospheric NO_x and N_2O from a fertilized agricultural soil, *Atmos. Environ.*, *25 Part A*, 1961-1969, 1991.

Shiklomanov, I.A., World fresh water resources, in *Water in Crisis: A Guide to the World's Fresh Water Resources*, edited by P.H. Gleick, pp. 13-24, Oxford Univ. Press, New York, 1993.

Singh, A., and V. Balasubramanian, Organic recycling in Asian agriculture, in *Organic Recycling in Asia*, p. 241-245, Food and Agric. Org., Rome, 1980.

Singh, G., R. Babu, P. Narain, L.S. Bhushan, and I.P. Abrol, Soil erosion rates in India, *J. Soil Water Conserv.*, *47*, 97-99, 1992.

Sinton, J.E. (Ed.), *China Energy Databook*, p. IV-76, Lawrence Berkeley National Lab., Berkeley, Calif., 1996.

Smil, V., *Carbon Nitrogen Sulfur*, 459 pp., Plenum, New York, 1985.

Smil, V., Nitrogen and phosphorus, in *The Earth as Transformed by Human Action,* edited by B.L. Turner II et al., pp. 423-436, Cambridge Univ. Press, New York, 1990.

Smil, V., Population growth and nitrogen: An exploration of a critical existential link, *Pop. Dev. Rev.*, *17*, 569-601, 1991.

Smil, V., *China's Environmental Crisis*, 257 pp., M.E. Sharpe, Armonk, N.Y., 1993.

Smil, V., *Nutrient Flows in Agriculture*, 54 pp., Rep. Project 2050, Washington, D.C., 1994.

Smil, V., *Cycles of Life*, 221 pp., Sci. Am. Libr., New York, 1997a.

Smil, V., Some unorthodox perspectives on agricultural biodiversity: The case of legume cultivation, *Agric. Ecosyst. Environ.*, *62*, 135-144, 1997b.

Smil, V., Crop residues: Agriculture's largest harvest, *BioScience*, *49*, 299-308, 1999.

Sprent, J., *The Ecology of Nitrogen Cycle*, 160 pp., Cambridge Univ. Press, New York, 1987.

State Statistical Bureau, *China Statistical Yearbook 1997*, 830 pp., Beijing, 1997.

Stevenson, F.J., *Cycles of Soil*, 300 pp., Wiley-Interscience, New York, 1986.

Subba Rao, N. S., *Biofertilizers in Agriculture*, 208 pp., Oxford Univ. Press, New York, 1988.

Sutton, M.A., C.E.R. Pitcairn, and D. Fowler, The exchange of ammonia between the atmosphere and plant communities, *Adv. Ecol. Res.*, *24*, 301-390, 1993.

Svensson, B.H., L. Klemendtsson, S. Simkins, K. Paustian, and T. Rosswal, Soil denitrification in three cropping systems characterized by differences in nitrogen and carbon supply, *Plant Soil*, *138*, 257-271, 1991.

Terman, G.L., Volatilization losses of nitrogen as ammonia from surface-applied fertilizers, organic amendments, and crop residues, *Adv. Agron.*, *31*, 189-223, 1979.

Trimble, S.W., Denudation studies: Can we assume stream steady state?, *Science*, *188*, 1207-1208, 1975.

Tsai, S.M., P.M. da Silva, W.L. Cabezas, and R. Boneti, Variability in nitrogen fixation in common bean (*Phaseolus vulgaris* L.) intercropped with maize, *Plant Soil*, *152*, 93-101, 1993.

United Nations Organization (UNO), *World Population Prospects: The 1996 Revision, Annex I: Demographic Indicators*, 253 pp., United Nations, New York, 1998.

United Nations Industrial Development Organization (UNIDO) and International Fertilizer Development Center (IFDC), *Fertilizer Manual*, 615 pp., Kluwer Acad., Norwell, Mass., 1998.

United States Department of Agriculture (USDA), *Improving Soils with Organic Wastes,* 157 pp., Washington D.C., 1978.

United States Department of Agriculture (USDA), *Agricultural Yearbook*, Washington D.C., 1998.

Valiela, I., G. Collins, J. Kremer, K. Lajtha, M. Gesit, B. Seely, J. Brawley, and C. Sham, Nitrogen loading from coastal watersheds to receiving estuaries: New method and application, *Ecol. Appl.*, *7*, 358-380, 1997.

van der Voet, E., R. Kleijn, and H.A.U de Haes, Nitrogen pollution in the European Union: Origins and proposed solutions, *Environ. Conserv.*, *23*, 120-132, 1996.

van Dijk, T.A., Appropriate management practices to minimize the environmental impact of applied fertilizers, in *Nitric Acid-Based Fertilizers and the Environment,* edited by R.G. Lee, pp. 349-358, Int. Fertilizer Develop. Cent., Muscle Shoals, Ala., 1993.

Van Leeuwen, E.P., G.P.J. Draaijers, and J.W. Erisman, Mapping wet deposition of acidifying components and base cations over Europe using measurements, *Atmos. Environ.*, *30*, 2495-2511, 1996.

Veldkamp, E., M. Keller, and M. Nunez, Effects of pasture management on N_2O and NO emissions from soils in the humid tropics of Costa Rica, *Global Biogeochem. Cycles*, *12*, 71-79, 1998

Vitousek, P.M., J.D. Aber, R.W. Howarth, G.E. Likens, P.A. Matson, D.W. Schindler, W.H. Schlesinger, and D.G. Tilman, Human alteration of the global nitrogen cycle: Sources and consequences, *Ecol. Appl.*, *7*, 737-750, 1997.

Warneck, P., *Chemistry of the Natural Atmosphere*, 757 pp., Academic, 1988.

Watt, B.K., and A.L. Merrill, *Handbook of the Nutritional Contents of Foods*, 190 pp., Dover, Mineola, N.Y., 1975.

Wedin, W.F., and T.J. Klopfenstein, Cropland pastures and crop residues, in *Forages*, vol. ii, edited by R.F. Barnes, D.A. Miller, and C.J. Nelson, pp. 193-206, Iowa State Univ. Press, Ames, 1995.

Weier, K.L., J.W. Doran, J.F. Power, and D.T. Walters, Denitrification and the dinitrogen/nitrous oxide ratio as affected by soil water, available carbon and nitrate, *Soil Sci. Soc. Am. J.*, *57*, 66-72, 1993.

Wetselaar, R. and G.D. Farquhar, Nitrogen losses from tops of plants, *Adv. Agron.*, *33*, 263-302, 1980.

Wetselaar, R., J.R. Simpson, and T. Roswall, (Eds.), *Nitrogen Cycling in South-East Asian Wet Monsoonal Ecosystems*, 216 pp., Aust. Acad. of Sci., Canberra, 1981.

Whitehead, D.C., and N. Raistrick, Nitrogen in the excreta of dairy cattle: Changes during short-term storage, *J. Agric. Sci*, *121*, 73-81, 1993.

Williams, E.J., G.L. Hutchinson, and F.C. Fehsenfeld, NO_x and N_2O emissions from soil, *Global Biogeochem. Cycles*, *6*, 351-388, 1992.

Zhu, Z., Nitrogen balance and cycling in agroecosystems of China, in *Nitrogen in soils of China*, edited by Z. Zhu, Q. Wen, and J.R. Freney, pp. 323-338, Kluwer Acad., Norwell, Mass., 1997.

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