Biogeochemistry **52:** 281–320, 2001.

Policy implications of human-accelerated nitrogen cycling

ARVIN R. MOSIER[1*], MARINA AZZAROLI BLEKEN[2], PORNPIMOL CHAIWANAKUPT[3], ERLE C. ELLIS[4], JOHN R. FRENEY[5], RICHARD B. HOWARTH[6], PAMELA A. MATSON[7], KATSUYUKI MINAMI[8], ROZ NAYLOR[7], KIRSTIN N. WEEKS[6] & ZHAO-LIANG ZHU[9]

[1]*USDA/ARS, Fort Collins, CO 80522, U.S.A.;* [2]*Agricultural University of Norway, Aas, Norway;* [3]*Thailand Department of Agriculture, Bangkok, Thailand;* [4]*Center for Agroecology and Sustainable Food Systems, University of California, Santa Cruz, CA, U.S.A.;* [5]*CSIRO, Canberra, ACT, Australia;* [6]*Dartmouth College, Hanover, NH, U.S.A.;* [7]*Stanford University, Palo Alto, CA, U.S.A.;* [8]*NIAES, Tsukuba, Japan;* [9]*Chinese Academy of Science, Institute of Soil Science, Nanjing, China (***author for correspondence, e-mail: amosier@lamar.colostate.edu)*

Received June 13 2000; accepted June 26 2000

Key words: fertilizer, food production, fossil fuel combustion, mitigation, N, NO_x, N_2O

Abstract. The human induced input of reactive N into the global biosphere has increased to approximately 150 Tg N each year and is expected to continue to increase for the foreseeable future. The need to feed (∼125 Tg N) and to provide energy (∼25 Tg N) for the growing world population drives this trend. This increase in reactive N comes at, in some instances, significant costs to society through increased emissions of NO_x, NH_3, N_2O and NO_3^- and deposition of NO_y and NH_x.

In the atmosphere, increases in tropospheric ozone and acid deposition (NO_y and NH_x) have led to acidification of aquatic and soil systems and to reductions in forest and crop system production. Changes in aquatic systems as a result of nitrate leaching have led to decreased drinking water quality, eutrophication, hypoxia and decreases in aquatic plant diversity, for example. On the other hand, increased deposition of biologically available N may have increased forest biomass production and may have contributed to increased storage of atmospheric CO_2 in plant and soils. Most importantly, synthetic production of fertilizer N has contributed greatly to the remarkable increase in food production that has taken place during the past 50 years.

The development of policy to control unwanted reactive N release is difficult because much of the reactive N release is related to food and energy production and reactive N species can be transported great distances in the atmosphere and in aquatic systems. There are many possibilities for limiting reactive N emissions from fuel combustion, and in fact, great strides have been made during the past decades. Reducing the introduction of new reactive N and in curtailing the movement of this N in food production is even more difficult. The particular problem comes from the fact that most of the N that is introduced into the global food production system is not converted into usable product, but rather reenters the biosphere as a surplus. Global policy on N in agriculture is difficult because many countries need to increase

food production to raise nutritional levels or to keep up with population growth, which may require increased use of N fertilizers. Although N cycling occurs at regional and global scales, policies are implemented and enforced at the national or provincial/state levels. Multinational efforts to control N loss to the environment are surely needed, but these efforts will require commitments from individual countries and the policy-makers within those countries.

Introduction

This paper provides a view of some of the complexities of release of newly fixed, reactive nitrogen and national and international environmental policy. To introduce the topic we briefly discuss some of the issues related to human-induced changes to the nitrogen cycle and how these changes relate to policy issues. We first look at some of the main concerns that result from introduction of reactive N into the biosphere through food production and fossil fuel consumption. We discuss some of the changes in N use distribution globally and the impact on reactive N production of changes in human diet. We also explore how the decoupling of cereal grain and livestock production systems contribute to increased reactive N production and changes in regional and global redistribution of reactive N. In the final section we discuss policy approaches related to N use in agriculture and in fossil fuel consumption.

Characteristics of change in the nitrogen cycle

Nitrogen regulates numerous essential ecological and biogeochemical processes, including species composition, diversity, population growth and dynamics, productivity, decomposition, atmospheric chemistry, and nutrient cycling of many terrestrial, freshwater, and marine ecosystems. While human activities have altered the N cycle in a number of ways, the most fundamental change is the dramatic increase in biologically available N, also termed 'reactive N'. Human activities have more than doubled the rate of transfer of N from the highly abundant but biologically unavailable form di-nitrogen (N_2) in the atmosphere to available forms such as ammonium (NH_4^+), nitrite (NO_2^-), and nitrate (NO_3^-) in the biosphere (Smil 1999; Vitousek et al. 1997; Galloway et al. 1995; Vitousek & Matson 1993).

Prior to extensive human alteration, the primary pathways for transfer from inert to available forms of N were biological N fixation by specialized bacteria (accounting for around 100 Tg y^{-1} in terrestrial ecosystems and 30–300 Tg y^{-1} in marine systems) and lightning fixation (accounting for up to 10 Tg y^{-1}) (Vitousek et al. 1997). A number of anthropogenic pathways have now more than doubled the amount of reactive N coming into the biosphere each year. These pathways include industrial fixation of N for use

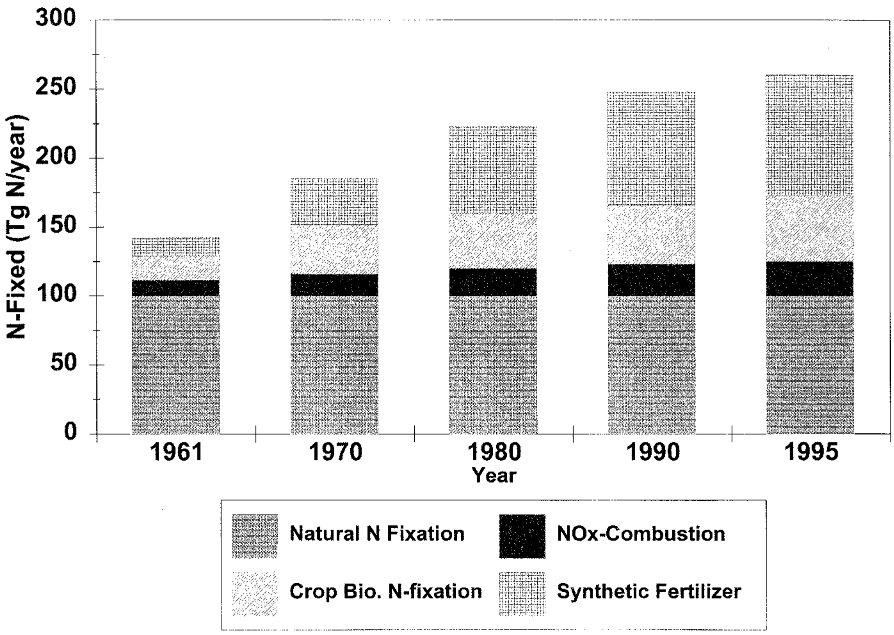

Figure 1. An estimate of global N fixation for 1961–1995 (FAO 1999; Galloway et al. 1995).

as fertilizers, cultivation of crops that fix N symbiotically, and mobilization and fixation during fossil fuel combustion, and have been the subject of a number of thorough reviews (e.g., Smil 1990, 1991, 1999; Vitousek & Matson 1993; Ayers et al. 1994; Galloway et al. 1995; Vitousek et al. 1997).

Current industrial fixation of N for fertilizer is approximately 85 Tg y^{-1}, a dramatic increase since 1961 (Figure 1). Until the late 1970s, most of the industrial N fertilizer was applied in developed countries, but use there has stabilized or declined while it has increased in developing countries (Figure 3). Animal manure and other organic residues are also used as fertilizer in crop production world-wide, and account for more N application than does industrial fertilizer (Bouwman 1997). Manure and organic residues represent recycling of already fixed N rather than new fixation of N; nevertheless, their management can have significant effects on the mobility of N and contribute to alterations in biospheric processes due to N use in human activities.

In addition to the use of fertilizers, agriculture has increased available N via the production of leguminous crops. Leguminous crops and forages, such as soybeans and alfalfa, support symbiotic N-fixing organisms, and their fixation rates typically far outstrip rates found in the natural systems that the crops replaced. Galloway et al. (1995) estimate that between 32 and 53 Tg N is fixed by crops annually.

The third major anthropogenic source of available N is via the burning of fossil fuels. Fossil fuel combustion inadvertently oxidizes atmospheric N_2 and also transfers some small amount of N from geologic reservoirs to available forms. Approximately 21 Tg N y^{-1} is converted to nitrogen oxides, which then react in the atmosphere and/or deposit to downwind ecosystems in gaseous, solution or particulate forms (Delmas et al. 1997).

Altogether, human activities cause the fixation of approximately 150 Tg N y^{-1} in terrestrial ecosystems, equivalent to biological N fixation by non-anthropogenic processes on land. In addition, land clearing, biomass burning, and drainage of wetlands all contribute substantially to the mobility of N in the biosphere, leading to reductions in long-term storage of N in soil organic matter and vegetation, and increasing fluxes to air and water.

These changes in available N, caused both directly and indirectly by human activities, affect the biosphere in many ways. They are associated with increased emission, transport, and deposition of a number of trace gases, including nitric oxide (NO) and ammonia (NH_3), both involved in air chemistry and downwind deposition, and nitrous oxide (N_2O), a greenhouse gas. Anthropogenic activities contribute significantly to all three of these gases. Deposition of nitrogen oxides, ammonia, organic N compounds, and other forms of N have consequences for downwind terrestrial and aquatic ecosystems, and may lead to increased production and decomposition, changes in species composition and biodiversity, and ultimately N saturation, with declining production and C storage and accelerated N losses (Aber et al. 1995, 1998; Galloway et al. 1995; Vitousek et al. 1997).

These anthropogenic changes also have led to major increases in N loading of aquatic systems over time. For example, nitrate concentrations in major rivers of the northeastern US have increased 3- to 10-fold since the early 20th century. Howarth et al. (1996) suggest that total riverine fluxes from most of the temperate zone land systems surrounding the North Atlantic have increased 2- to 20-fold since pre-industrial times. These elevated nitrogen concentrations and fluxes hold human health concerns due to pollution of drinking water, and also affect downstream ecosystems through acidification and eutrophication (Vitousek et al. 1997).

Considerations for policy development

Clearly, the dramatic changes in the global N cycle outlined above hold significant consequences for the way the Earth system functions. Increased nitrogen availability at the global scale has affected and will continue to affect terrestrial and aquatic systems alike. Some of those consequences are playing out as air (ozone formation due to NO_x emissions from soils) and water pollution problems (enrichment in nitrate from leaching and runoff from cropped field and domestic lawns) at local scales; others are reflected in regional changes in net primary production and eutrophication; still others are global in scale. As the source of multi-dimensional environmental problems, global change in the N cycle is drawing interest among managers and policy makers. There are, however, a number of characteristics and dynamics of this change that make the problems stemming from global change in N especially difficult to solve.

The first and most central characteristic to be considered in policy development is the fact that N use is closely tied to essential human endeavors such as the provision of food and energy. Fertilizer use has been an essential component of the Green Revolution, the set of technologies that dramatically increased food production in developing countries during the period between 1960 and 1980. Many areas of the world still do not use enough fertilizer to maximize crop yields, and most analysts suggest that fertilizer use will continue to grow as food production does, in order to keep pace with a still-rapidly increasing human population. Use of N for agriculture, whether in the form of synthetic or organic fertilizer, is not substitutable, and straightforward technological changes are unlikely to provide a replacement. Plants will always require a relatively large amount of N to carry out their photosynthetic processes, and one key to maintaining adequate food supplies will be supplying plants with adequate N (Matson et al. 1997). This characteristic sets fertilizer N apart from those environmental problems that can be solved by technological substitutions; for example, chlorofluorocarbon (CFC)-caused reduction in stratospheric ozone is being dealt with effectively by substituting non-ozone depleting chemicals in the myriad industrial processes that in the past relied on CFCs.

Global change in the N cycle is also linked to the use of fossil fuel energy, and thus is likely to increase dramatically over the next several decades, unless there is a concerted effort to control fossil fuel consumption. Galloway et al. (1994) suggest that the production of NO_x from fossil fuels will double over the next several decades, reaching approximately 46 Tg y^{-1} by 2020. In this case, technological change that either increases efficiency of fuel combustion or removes nitrogen oxides from the exhaust stream could reduce the total amount of N emitted, but complete solutions are closely linked to the development of non-polluting alternative energy sources.

The second characteristic of N that is critical to policy making is the fact that N is a highly mobile element – as one publication pointed out, it seems to ‘hopscotch’ around the globe (Galloway et al. 1995), moving through air and water, across political and geographical boundaries. As a result, sources and sinks (or cause and effect) are often widely separated. Thus, eutrophication in the Gulf of Mexico is linked to fertilizer use in the

Mississippi Valley (Downing et al. 1999), and N deposition and acid rain in Scandinavia are linked to fossil fuel burning and agriculture in nations to the south (Abrahamsen & Stuanes 1998). One consequence of this mobility is that policies to solve environmental problems associated with it must often be multi-national in scale.

Finally, a third characteristic of N that affects policy making and management approaches is the fact that changes in N are interactive with other global changes. To completely understand the effects of N additions to ecosystems, one must understand how those additions interact with elevated CO_2, with land use change, with biological invasions, and with other biogeochemical changes. For example, attributing forest dieback to N deposition alone has been quite difficult, because many forests are also being subjected to increased exposure to tropospheric ozone. Likewise, attributing increased forest growth to N deposition is complicated by the fact that climate change and elevated CO_2 are happening simultaneously. No policies to date are comprehensive enough to address the multiple and interacting changes that are occurring globally.

Nitrogen use in food production

Trends in fertilizer use and food production

Since 1950, N input into global crop production has greatly increased as have crop production and human population. In 1950 synthetic fertilizer N input comprised $\sim$7% of total N input of $\sim$56 Tg N. In 1996 synthetic N input was $\sim$43% of the total N input (including biological N-fixation in crops) of 190 Tg for global crop production (FAO 1999; IPCC 1997). Animal waste used as fertilizer was an estimated 37 Tg in 1950 compared to $\sim$65 Tg N in 1996. Globally, synthetic fertilizer N consumption is expected to grow relatively constantly at the rate of 1.6 Tg N y^{-1} between 2000 and 2020 (Bumb & Baanante 1996). In 1961 FAO began compiling world population, crop production and fertilizer use statistics (FAO 1999) and since that time global human population increased from $\sim$3.1 billion to $\sim$5.8 billion in 1996 (a 1.9 fold increase and an annual growth rate of $\sim$2.5% y^{-1} using 1961 as the base time) (Figure 2). During this time world cereal grain production increased from $\sim$880 Tg to $\sim$2070 Tg, a 2.5 fold increase representing an annual gain of over 4.1% y^{-1}. Synthetic fertilizer N consumption increased from 11.6 Tg to almost 83 Tg in 1996. This 7.1-fold increase in fertilizer N use was the result of very rapid expansion in use between 1961 and 1980 (an annual increase of $\sim$22%).

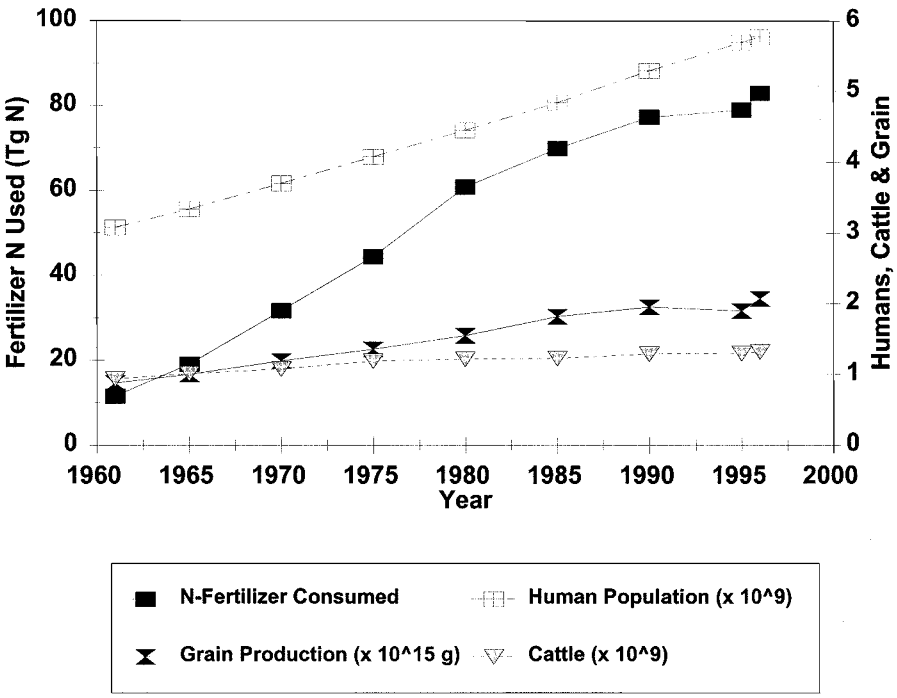

Figure 2. World human population, grain production, N-fertilizer consumption and cattle population (FAO 1999).

The continued increase in food production globally over the past 35 years has been remarkable. Particularly, since land area used to produce grain decreased from 0.2 to 0.12 ha per person between 1965 and 1994 while global grain production per person increased 16%. Agricultural production increase has come through the use of new crop varieties which need increased N-fertilization, pesticide use, irrigation and mechanization. About 40% of the large annual increase in crop production during this 35-y period is attributed to the increase in use of synthetic fertilizer N (Brown 1999).

Even though agricultural production has increased dramatically, fertilizer N use efficiency remains relatively low. Crops typically take up only 40–50% of the total organic and inorganic N added during each cropping season (Bleken & Bakken 1997; NRC 1993; Olsthoorn & Fong 1998). Since fertilizer N is not used efficiently in most parts of the world, N use in excess of crop requirements leads to losses to the environment through volatilization and leaching. Improved efficiency of fertilizer use can be attained, at least in part, through improved management, using current technology, to reduce N input into cropping systems without decreasing production (Peoples et al.

1995; Cole et al. 1996; De Jager et al. 1998; Hendriks et al. 1998; Matson et al. 1998; Mosier et al. 1998; Downing et al. 1999)

Shifts in global use of fertilizer N and crop production

The need for continued improvement in management of N in crop production is further enhanced by the changes in the global distribution of N-fertilizer use in the past few decades. Management techniques may or may not be applicable to the vast array of crop production systems in use globally, since much of the research on fertilizer N management techniques has been conducted in the developed part of the world. According to the FAO terminology of developed and developing countries (FAO 1999), fertilizer N use and grain production have declined in developed countries since 1985 (Figure 3). Fertilizer N consumption peaked in the developed part of the world in 1985 at 38.5 Tg and dropped to 28 Tg in 1994. The data are not shown here, but a close look at the use of N in developed countries shows that the decrease is due mainly to lower consumption in Eastern Europe and the former Soviet Union (FAO 1999). Since 1995 N use in North America and other parts of the developed world began to increase again. In the developing part of the world both grain production and N-fertilizer use have continued to increase at near linear rates over the past three decades. Cereal grain production increased from $\sim$400 in 1961 to $\sim$1200 Tg in 1996 while fertilizer N consumption increased from 2.2 to $\sim$53 Tg (a 24 fold increase). On a per capita basis fertilizer N use in the developing world increased from $\sim$1.1 kg N y^{-1} per person in 1961 to 11.6 kg N y^{-1} per person in 1996. During the same period human population in the developing world increased from $\sim$2.1 billion to $\sim$4.6 billion.

Human population in the developed world increased from 0.98 to 1.29 billion while grain production increased from 481 in 1961 to 913 Tg in 1990. In 1996, 1997 and 1998 developed country grain production was 867, 906 and 856 Tg, respectively (FAO 1999). On a per capita basis fertilizer N use in the developed part of the world was 9.6 and 23.3 kg N y^{-1} in 1961 and 1996, respectively.

Increases in N loss resulting from changes in human diet and food production systems

Increased N demand due to changes in human diet

Along with increasing fertilizer N use, continued high intake of animal protein in developed countries and changes in the diet of people in developing

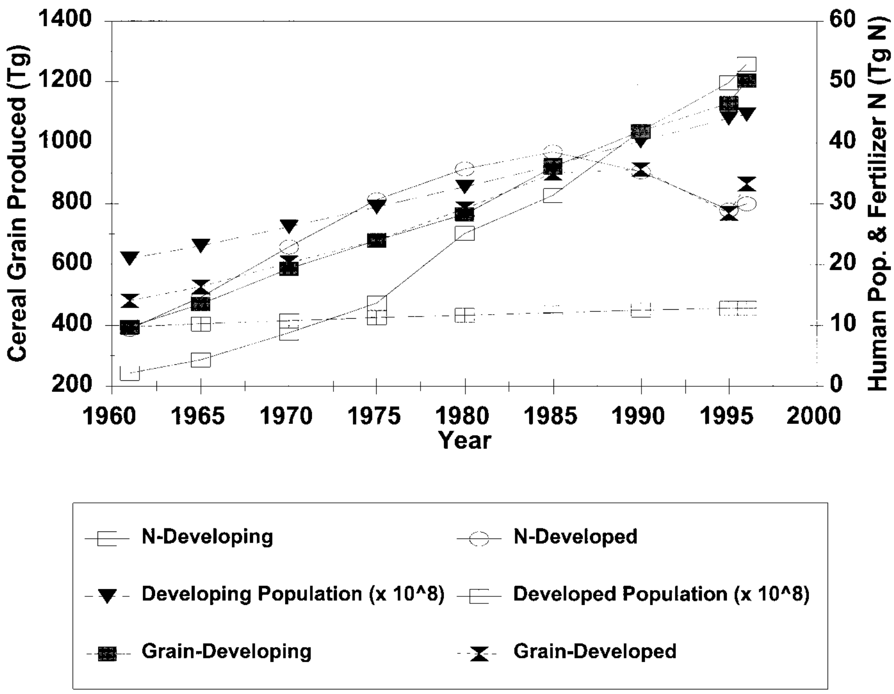

Figure 3. Distribution of N fertilizer use, grain production and human population in developed and developing parts of the world (FAO 1999).

countries will likely lead to greater N losses from global food production in the future. The first aspect of changes in food production concerns increasing meat consumption by people globally and resulting need for increased N input into food production (Figures 3, 4, 5 & 6).

In the developed part of the world total meat and cereal grain production were high in 1980 and have increased slightly since that time (Figures 3 & 4). In contrast, in the developing world, total meat production increased from ∼40 Tg in 1980 to more than 110 Tg in 1998 (Figure 4). Grain production increased from ∼800 to ∼1200 Tg during this time (Figure 3). Improved economic conditions, particularly in East Asia, appear to have stimulated meat consumption at a rate greater than population increase (Figures 3 & 4). Even with the very rapid increase in meat production in the last decade, per capita protein consumption in the developing world remains much lower than in developed countries. Badiane and Delgado (1995) note that increased animal production may be necessary in countries that are poor and have a large rural population, as in Sub-Saharan Africa, to meet food needs and for the economic development of the country.

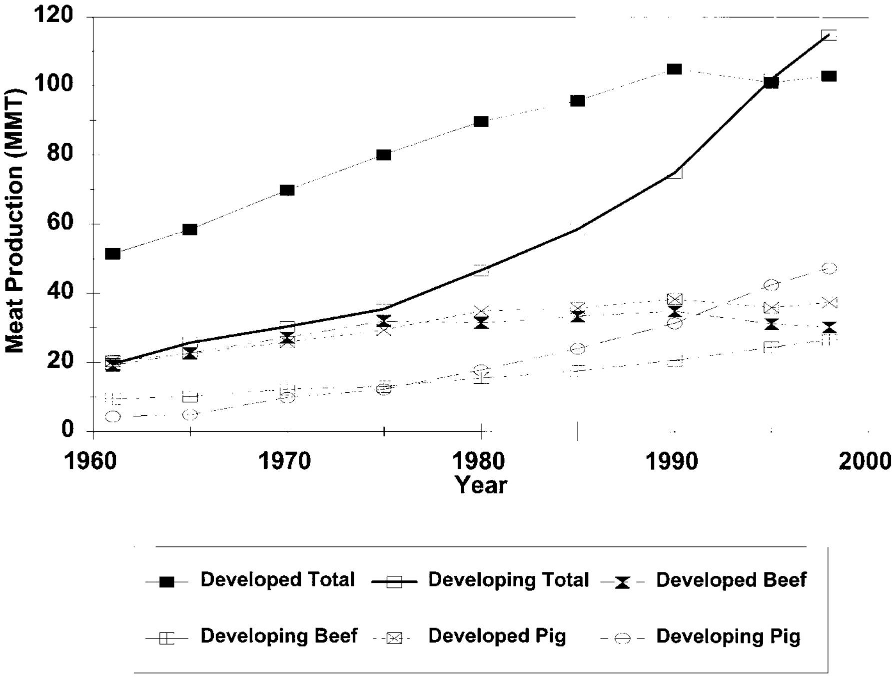

Figure 4. Changes in meat production since 1961 in developed and developing countries (FAO 1999).

The average protein supply per person in the developed countries is presently $\sim$100 g d^{-1}, while in the developing countries it is only $\sim$65 g d^{-1} (Figure 5). Protein is used because there is a direct proportionality between protein and nitrogen composition of food (ca 0.16 g N per 1 g protein). On average in 1995, developed countries consumed $\sim$55% of total protein from animal sources while developing countries derived $\sim$25% of total protein from animals (Figures 5 & 6). Protein consumption was highest in the USA and Western Europe, $\sim$70 and $\sim$60 g animal protein person^{-1} d, respectively. Protein consumption in the Former USSR declined markedly during the past decade, mainly due to the decrease in animal protein. In developing countries, the greatest change in animal protein consumption has occurred in China where the consumption of meat products has increased 3.2 fold since 1980. In Sub-Saharan Africa there has been no increase in either total or animal protein supply during the past 30+ years (Figures 5 & 6).

The reason for focusing on the consumption of animal protein is that more N is needed to produce a unit of animal protein than an equal amount of plant protein. This implies greater N loss from the soil than if crops were used directly as human food. Using information from Western Europe, Bleken and

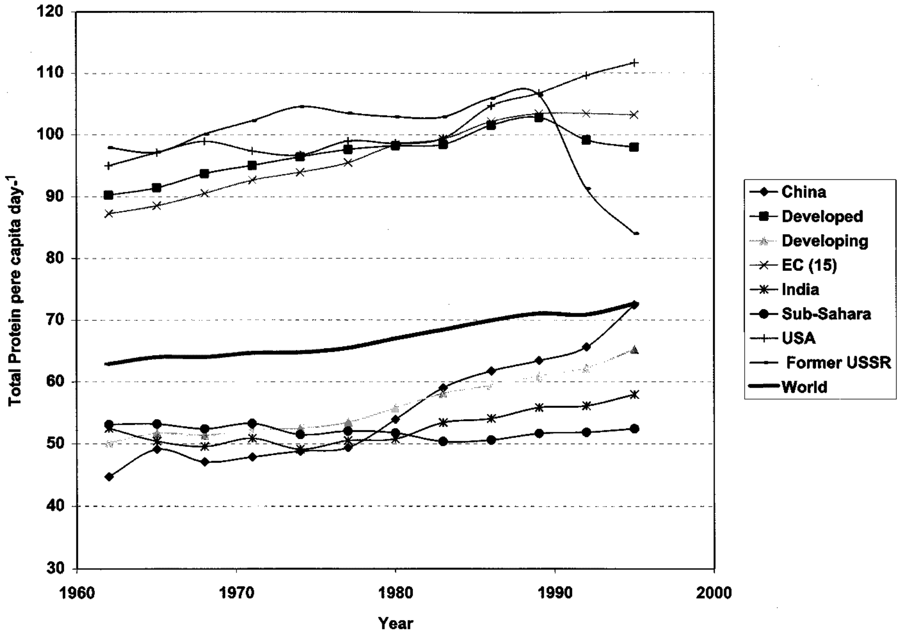

Figure 5. Human consumption of total protein in various parts of the world (FAO 1999).

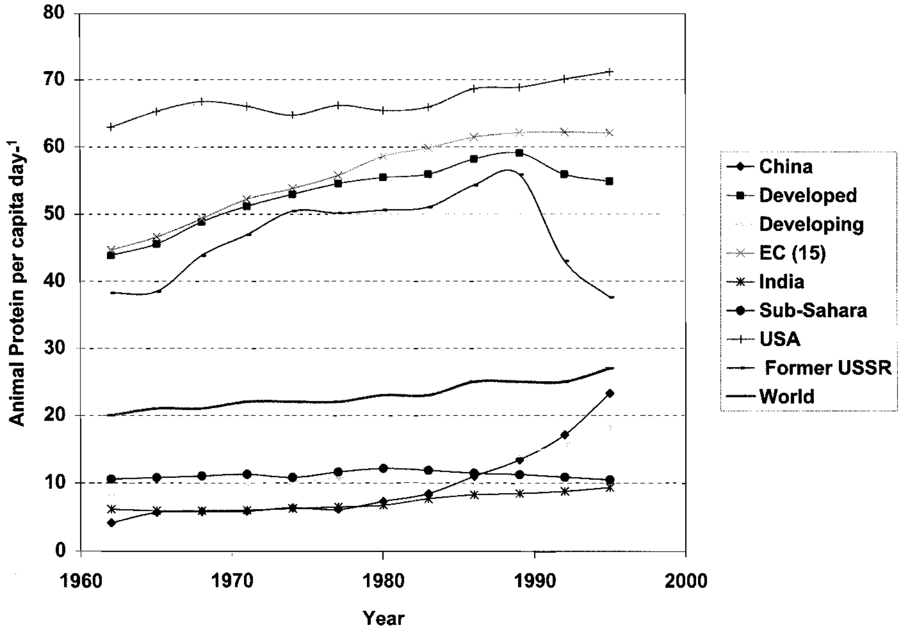

Figure 6. Human consumption of animal protein in various parts of the world (FAO 1999).

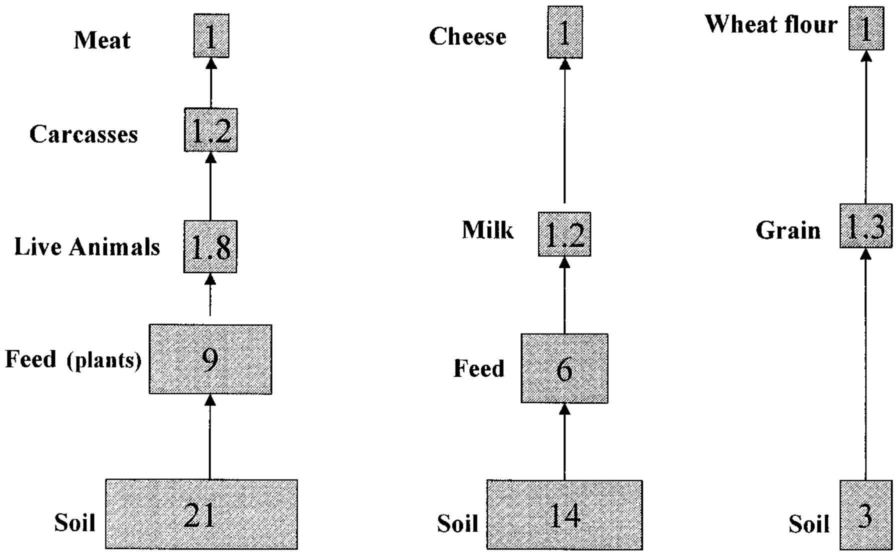

Figure 7. A diagram of flow of reactive N from application to soil through intermediary plant and animal products to produce a unit of edible protein N. The numbers indicate of amount of N. One g N corresponds to ~6.25 g protein, ~35 g meat, ~200 g milk or ~50 g wheat flour (Bleken & Bakken 1997).

Bakken (1997) found that about 3 g N must be supplied to the soil (sum of fertilizer, biological fixation and atmospheric deposition) to produce wheat flour containing 1 g N (~6.3 g protein). In order to obtain an equal amount of animal protein (weighted average of milk and several kinds of meat) the soil must receive 21 g N. Taking into account recycling of manure and recycling of animal products as feed, the net external N input to the agricultural system necessary to produce this amount of edible animal proteins is ~16 g N (Figure 7). This is more than five times the requirement for production of vegetable protein. This illustrates the fact that even though part of the N in animal feed can be recycled for fertilizer in crop production as animal manure, large losses occur from volatilization of NH_3 and from nitrate leaching and runoff and N_2O emissions following mineralization. Based upon the extra N required to produce animal protein (Figure 7), continued high animal protein consumption in developed countries and changes to animal protein-based diet in developing countries will likely increase N input and N losses into food production.

Moderating this increase by decreasing the average amount of animal protein consumed in developed countries is one mechanism of limiting part of the expected increased N requirement in food production. Bleken (1997) analyzed the relation between human diet and global N need for food production. Her analysis indicates that the total N input for diets with high animal protein intake is almost twice as high as the N needed for diets of moderate animal protein. Her analysis was based on a historical development of food consumption for different countries based on FAO (1999) data. One example of a country with a good food supply and moderate consumption of animal protein is Italy in 1963. At that time food supply was adequate to ensure sufficient nutrition to all groups of society (Bleken 1997). Total protein consumption was 85 g per capita d^{-1}, and consumption of animal protein was 32 g. This is considerably lower than the present average of the developed countries, and yet much higher than the average of developing countries. Another example is Japan, where animal protein consumption has traditionally been low, although it has increased from 25 g in 1963 to 54 g animal protein per capita d^{-1} in 1995. In the same period the total protein consumption has increased from 73 g to 96 per capita d^{-1}.

Increases in N loss due to the decoupling of livestock and crop production

The increase in consumption of animal products worldwide has been accompanied by an intensification of animal production in limited regions. This production localization is generating a second important contribution to increased N loss that results from the disassociation of crop and livestock production. In some European countries livestock feed is imported from other continents (Bouwman & Booij 1998). This spatial intensification of livestock production has caused a high concentration of animal manure on little land, with the consequence that manure has been regarded as waste rather than as a plant nutrient resource.

The relationship of livestock and crop production in the U.S. is an example where the dietary consumption of meat has been at a relatively high level for many decades. As a result of domestic consumption and exports, U.S. livestock production has historically been a large part of U.S. agricultural production. Meat production in the U.S. (beef, pork, poultry, lamb etc.) increased from about 90 kg/person in 1963 to about 125 in 1995 (FAO 1999). During this time there has been a shift in the location of U.S. livestock production, mainly to the Southern and Atlantic states from the Midwest and Great Plains. Much of this shift in resulting N deposition can be attributed to the increase in poultry and hog production in the South and Atlantic coastal sections of the country. There were increases in livestock production of 140% in the North Atlantic region, 270% in the South Atlantic region, 160% in the Midsouth and 140% in the Southern Plains from 1963 to 1995. At the same time livestock production decreased 40% in the Midwest and 30% in the Northern Plains where more than 70% of U.S. maize and wheat is produced (USDA 1997).

Table 1. Nitrogen balance sheet for global agriculture in 1994 (Van der Hoek 1998).

N input	Tg N	Products	Tg N
Fertilizer	74	Animal	12
N-fixation	50	Crops	40
Feeds	10	Surplus[1]	90
Unaccounted	10		
Total	140	Total	140

[1] Surplus is defined as the difference between input and output in animal or crop products.

Because of the centralization of livestock production into regions that produce relatively little animal feed, the area of crop production located in close proximity to the intensive animal production systems are not adequate to carry the animal waste input load. This disassociation between animal and crop production generates transportation costs for the manure that are greater than the N or other nutrient value of the waste (NRC 1993). Of the $\sim$11 Tg N in animal waste excreted in the U.S. in 1990, an estimated 34% was returned to cropped fields for use as fertilizer (NRC 1993). The remainder of the N is stored in lagoons or solid piles (Smil 1999) or distributed elsewhere partly through NH_3 volatilization (an estimated 25% of N excreted in an open cattle feedlot is directly volatilized as NH_3) (Hutchinson & Viets 1969) surface runoff, leaching and wind erosion. Most of the volatilized NH_3 is deposited near the feedlot but significant amounts can be converted to aerosols and transported 1000+ km (Ferm 1998). Much of the remaining ‘unused’ N eventually finds its way into ground and surface waters (Downing et al. 1999).

Van der Hoek (1998) estimated that 63% of the annual N input into food production was not converted into usable product. This ‘surplus N’, defined as the difference between input and output, is either lost to the environment or accumulates in the soil (Table 1).

Increased N loss resulting from changes in crop production techniques

Traditionally, crop production, animal production and human consumption were closely linked geographically in many parts of the world. Countries, such as China, have long histories of intensive human and animal food production that was conducted such that livestock and human wastes were returned to the field. This practice kept the N cycle more tightly linked within the food production system, thus minimizing N losses and maintaining soil fertility. In the past few decades the increased demand placed on food production from growing human populations coupled with increased consumption of animal protein has led to significant changes in crop production systems. N input into crop production from synthetic fertilizer has increased while N from animal waste has become relatively less important. This aspect is important for P and K and minor nutrient supply and maintaining soil organic matter (Zhu 1998a).

An important example is the current trend of livestock production in China where, based on data from each Province (Almanac of Agriculture in China 1982, 1997), the ratio of animal production (number of cattle + pigs + sheep) to that of cereal grain (rice + maize + wheat) produced increased from 0.82 to 1.58 between 1981 and 1996. Thus animal production was increasing more rapidly than cereal grain production. Even though livestock production is increasing, utilization of animal manures for fertilizers appears to be on the decline. To illustrate this point we use two examples from China. The first example is highlighted in Box 1 where changes in N management by Chinese villages in the Tai Lake region of China are documented.

Jiangsu Province example. The second example comes from Changshu in Jiangsu Province (Figure 8). Manure N input decreased from 50 to 20 kg N ha^{-1} between 1986 and 1997 while synthetic fertilizer N input increased from 113 to 274 kg N ha^{-1}. Total N input increased from 160 to 290 kg N ha^{-1}, an 80% increase while total grain yield ha^{-1} increased only 8%. Within Jiangsu Province between 1981 and 1996, the number of cattle, pigs and sheep sold increased by 83, 50 and 595%, respectively (Almanac for Agriculture of China 1982, 1997). With less manure N being used for crop production and more livestock being grown, where is the increased animal waste N going? A nutrient budget analysis of a large watershed in Southeastern China by Yan et al. (1998) suggests that much of the excess N is moving into aquatic systems. The nitrate content in the water of rivers and lakes in regions that have high N input and high crop production and has increased several fold during last 10 years (Zhu 1998a; Zhuang et al. 1995), due to the high losses of N from croplands (Zhu 1997, 1998b; Li et al. 1998).

Options for limiting N input into crop production

It is clear from many reports that when fertilizer N is applied in an amount needed by the crop for near optimum production, and at the time that the plants use the N, that N losses are relatively small. As an example, Broadbent and Carlton (1978) conducted a 3-year series of fertilizer N utilization trials in irrigated maize using ^{15}N-depleted ammonium sulfate fertilizer. Their study on a sandy loam soil in central California was supplemented with leaching, Since 1949, fertilizer N inputs have increased more than five-fold in the densely populated alluvial flood plain of the Tai Lake Region. For centuries, farmers used less than 100 kg N ha^{-1} y^{-1} from legume green manure, canal sediments, manure and crop residues to sustain 4 mg ha^{-1} rice yields in their N-limited rice/wheat double cropping systems (Ellis & Wang 1997). Starting in the 1960s, synthetic N applications began to intensify, pushing paddy fertilizer inputs to nearly 500 kg N ha^{-1} y^{-1} by the late 1980s, doubling rice yields in support of doubled human populations. Over the same period, nitrate pollution and eutrophication became common (Ma 1997). By integrating household and landscape data collected on-site from 1993 to 1996 with historical data from reference materials and village elder interviews, the long-term impacts of these developments are being reconstructed in a typical floodplain village of the region. This village-scale analysis helps identify both the sources of large-scale environmental problems and pathways toward their solution. (Ellis et al. 2000a; Ellis et al. 2000b).

Synthetic N increased from 0% of paddy fertilizer N in 1930 to more than 80% in 1994, displacing traditional organic inputs and altering N storage and cycling. When communal agriculture ended in 1982, synthetic N replaced the traditional labor-intensive practice of harvesting canal sediments for fertilizer. Short of a political intervention, these unharvested sediments will completely fill most village canals within 25 years, increasing flood risk and impeding irrigation and transport. Another traditional fertilizer, nightsoil (human manure), was applied mostly to paddy land in the past, at rates rarely exceeding 40 kg N ha^{-1} y^{-1}. Now, to save labor, most nightsoil is applied to small upland plots near houses at rates greater than 200 kg N ha^{-1} y^{-1}, transforming an essential fertilizer into an excess nutrient source in the drier areas of the village most susceptible to nitrate leaching. Overall, changes in N management and land use have increased N storage in village soil and sediment by about 25% from 1930 to 1994, or by 1.4 mg N ha^{-1} on average. Accumulation in sediment accounts for about half of the N buildup, with the remainder due to a 20% increase in agricultural soil N concentration caused by the intensive use of N fertilizers.

Sustaining high yields is critical for village food security. Paddy land availability, already low in 1930 (0.11 ha $person^{-1}$) was halved to just 0.051 ha $person^{-1}$ by 1994 and is still declining. Annual paddy N inputs to rice/wheat systems averaged 480 kg N ha^{-1} y^{-1} in a 1994 sample of 50 village households, varying between 210 and 850 kg N ha^{-1} y^{-1}. As inputs greater than 500 kg N ha^{-1} y^{-1} do not increase yields, N inputs by nearly half of village households exacerbate environmental pollution without any possible benefit to farmers. Paddy N losses could be reduced significantly if only the 48% of households with above average N loading reduced their inputs to the average.

Box 1. Changes in N management by villages in the Tai Lake Region of China.

denitrification and modeling efforts. Their results and synthesis of these results by Legg and Meisinger (1982) show that maximum N use efficiency was found at the same fertilizer rate needed to obtain maximum yield. When N was applied in excess of this amount, large amounts of nitrate accumulated in the soil profile that were subject to leaching. Nitrification/denitrification losses (N_2 + N_2O + NO_x) were estimated to be a relatively constant $\sim$22% of N applied.

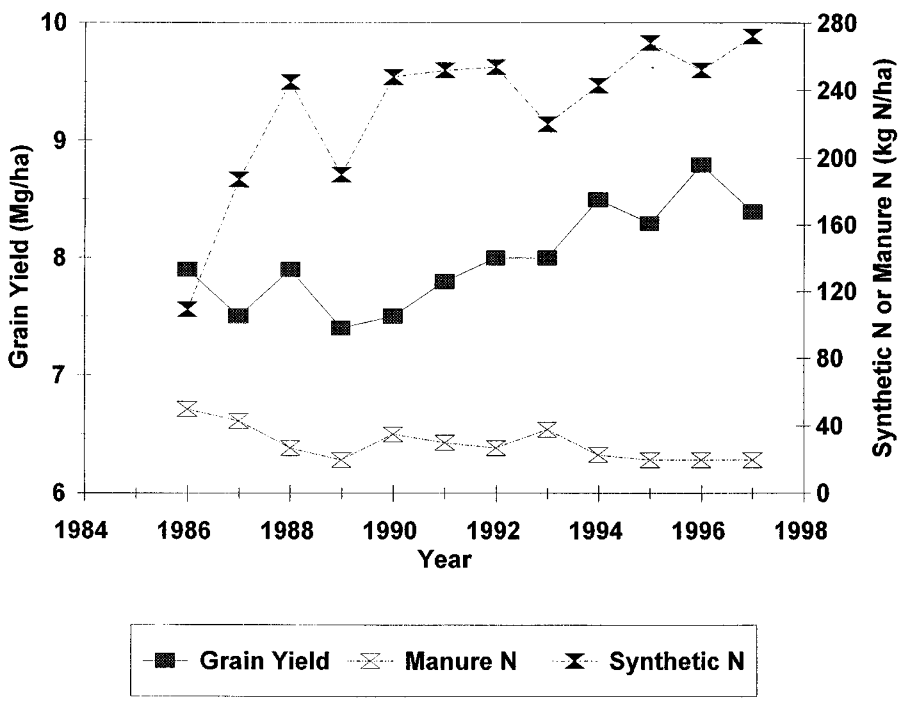

Figure 8. Animal manure N and synthetic N input into grain production in Changshu, Jiangsu Province, China (Zhu, Z.L., unpublished data).

These studies also showed that significant N losses through denitrification and leaching can be expected even at 'optimal' N application rates. The potential to conserve almost one third of the fertilizer N plus mineralized soil N still exists for most cropping systems. Management techniques to limit N input while maintaining crop production need to continue to be incorporated into crop production systems. Some management options that have proven useful in reducing N input into cropping systems without decreasing production follow (Peoples et al. 1995; Cole et al. 1996; De Jager et al. 1998; Hendriks et al. 1998):

- Match N supply with crop demand by: Using soil/plant testing to determine fertilizer needs; optimizing split application schemes; or matching N application to reduced production goals in regions of overproduction
- Tighten N flow cycles by: Integrating animal and crop production systems to utilize animal manure in crop production; maintaining plant residue N on the production site; or minimizing fallow periods to limit mineral N accumulation

- Use advanced fertilization techniques such as: Controlled release fertilizers; place fertilizers below the soil surface; foliar application of fertilizers; nitrification inhibitors; or match fertilizer type to seasonal precipitation
- Optimize tillage, irrigation and drainage
- Optimize animal protein in human diet

The first option aims at a more efficient use of N in agriculture and generally applies to intensively managed types of agriculture. These strategies include a better matching of N supply with crop demand and optimizing split application schemes. An example of a system where a change in fertilizer management decreased N demand, decreased gaseous N losses and improved profit to the farmer is provided by Matson et al. (1998) from their study in irrigated wheat production in west-central Mexico. This study shows that although it is unlikely that losses can be completely eliminated that management can improve efficiency. The amount of fertilizer N required to produce the same crop was decreased 28%, thereby, decreasing N losses (nitrate leaching, NO_x and N_2O emissions) and increasing farmer profit. In areas of high input agriculture it has been assumed that fertilizer use efficiency could be improved by 20% (Cole et al. 1996; Peoples et al. 1995).

A combination of these first mitigation options are being considered for improving N utilization in China (Zhu 1998a): (1) to maintain the high yield level with a reduced rate of N application through improvement in N application techniques. Three mechanisms to promote this strategy are: (a) to optimize the rate of N application (Zhu 1997, 1998b). For example, in set of field experiments for flooded rice conducted in Wujin county, south Jiangsu province, Cui et al. (1998) found that the optimal rate of N application for maximum economic benefit and minimum leaching loss was 220–260 kg N ha^{-1} for a grain yield of around 7.5 t ha^{-1}, (b) to optimize timing of N application, by applying little N in the early stage of growth and to top-dress the major portion of fertilizer N at vigorous growth stages. Many ^{15}N-microplot field experiments have found that the N loss in from a late application was much less than from an early application (Zhu 1997) and (c) to deep-place the fertilizer N into the soil, which can reduce ammonia volatilization substantially (Zhu 1997). (2) A second choice is to reduce the yield target in the high-yielding regions. This assumes that the yield loss can be compensated by the yield increase in other regions of low and moderate productivity where the rate of N application is less than 150–180 kg N ha^{-1} $crop^{-1}$. This strategy could only be adopted by policy-makers if the increase in yield in the low yielding regions can be realized through improvement in irrigation, land leveling and the increase in the rate of N application and balanced fertilization (P and K additions in addition to N).

The second type of fertilization management option is applicable to most areas of the world. As we have shown in examples in China and the USA, the tightening N flow cycles through reintegration of animal and crop production systems could be used to decrease the demand for synthetically fixed N. Utilizing plant residue in N recycling to both increase soil fertility but also to reduce erosion could be increased in many parts of the world (Lal et al. 1998). Tightening N cycles could reduce the need for synthetic N input by another 10 to 20% if fully utilized (Cole et al. 1996).

The third and fourth options for decreasing N losses from agriculture include new technological advances and management. The development of control release fertilizers which can release fertilizer components at a rate that matches crop demand while protecting the remainder of the fertilizer from release (Minami et al. 1994; Shoji & Kanno 1994). Managing N input so that the fertilizer type applied helps decrease N loss. For example, in a seasonally wet pasture applying an ammonium-based fertilizer during the wet period and a nitrate-based fertilizer during the dry period significantly decreased N losses (McTaggart et al. 1997). Full use of these techniques have the potential to improve N utilization globally by 10–20% (Cole et al. 1996).

Finally, a change in the human diet may be an effective way to reduce the total amount of N cycled through the agricultural system. Bleken (1997) analyzed the relation between N needed for food production and the daily consumption of animal proteins. Her analysis indicates that total N needed for diets with high animal protein intakes (comparable to many industrialized countries today) are almost twice as high as the N needed for the average diets in Italy 1963 or Turkey 1993. Based on her analysis, we assume that in high-N input regions per capita N need for food production may be reduced by 45%, which would reduce present-day N inputs by about 15% worldwide.

Fossil fuel combustion and NO_x

Trends in NO_x emissions

Fossil fuel combustion is a major source of NO_x inputs to the atmosphere, accounting for worldwide emissions of some 21 Tg N in 1985 (Benkovitz et al. 1996) (Table 2). The source of these emissions may be disaggregated into two components. Thermal NO_x is generated by the oxidation of diatomic nitrogen as a by-product of combustion. Fuel NO_x is formed when the nitrogen contained in the organic compounds that comprise fossil fuels is released to the atmosphere. Although thermal NO_x is dominant for fuels with low nitrogen content such as natural gas and petroleum distillates, fuel NO_x accounts for 50–90% of the emissions associated with heavier fuels such

Table 2. Worldwide NO_x emissions from fossil fuel combustion in 1985 (Benkovitz et al. 1996).

	Tg N yr^{-1}
North America	6.24
Former USSR (stationary sources)	0.50
Europe, Middle East, North Africa	6.07
South Africa	0.22
Australia	0.21
25 Asian countries	4.20
Rest of world	3.56
Total	21.00

as residual oil and coal, which typically contain between 0.3% and 3.0% nitrogen by weight (Seinfeld 1986). Both pathways are strongly affected by combustion technologies and the details of equipment operation.

Estimating the anthropogenic emissions of NO_x that are linked to fossil fuel consumption is an imprecise art. Emissions are often not subject to direct measurement but are instead inferred from data on fuel use. In the simplest case, the consumption of various energy carriers (e.g. hard coal, lignite, gasoline, residual fuel oil, natural gas, etc.) is multiplied by average emission coefficients that are derived from field observations and/or laboratory studies (Müller 1992). This approach is useful in generating order-of-magnitude emission estimates, but is unable to account for the role of specific technologies in mediating the relationship between fuel use and NO_x emissions. This method, however, is often the best that can be done to estimate NO_x emissions in developing countries, where disaggregated data on the disposition of fuel consumption by end use or process are often of low quality or are entirely lacking.

Relatively detailed data are available for the advanced industrial nations of North America, Europe, Oceania, and Japan, where pollution control has been a long-standing policy priority. In the United States, for example, emissions of NO_x from stationary and mobile sources are regulated under the Clean Air Act, the main provisions of which date to 1970. This statute requires the monitoring of emissions from power plants and industrial facilities as well as detailed estimation of the emissions generated by automobiles and commercial vehicles. Given their role in the process of acid deposition, NO_x emissions have come under close scrutiny in both North America and Europe, where emissions abatement is mandated under the Convention on Long-

Range Transboundary Air Pollution. According to Bergesen and Parmann (1997), 21 out of 25 nations that signed onto this agreement have met their commitment to stabilize NO_x emissions at 1987 (or, for the United States, 1978) levels. These targets have been achieved using a combination of economic incentives and technology-based standards.

At a global level, NO_x emissions have grown exponentially throughout the 20th century (Holland & Lamarque 1997). Although emissions dipped briefly during the Great Depression and the energy shocks of the 1970s, they grew from near zero in 1900 to over 20 Tg N in the 1990s. According to the Global Emissions Inventory Activity project (Benkovitz et al. 1996), worldwide NO_x emissions in 1985 were distributed between regions as shown in Table 2.

Unfortunately, time series data on global NO_x emissions are not readily available from official sources, although the Organization for Economic Cooperation and Development (1997) publishes periodic statistics on pollutant emissions for member countries, which include the industrial democracies of North America, Europe, Oceania, and Japan.

Options for decreasing NO_x emissions

A range of approaches to NO_x emissions control has been implemented. At the most general level, emissions may be reduced by curtailing energy use, by switching from fossil fuels to alternative energy sources, or by altering the technologies through which fossil fuels are harnessed to provide heat and power. It is well-recognized that energy use is an important contributor to human well-being, providing the services necessary to heat and cool buildings, operate vehicles and equipment, provide lighting, and drive industrial processes. Since the 1970s, however, major industrial economies have attained substantial economic growth without commensurate increases in energy use (Schipper & Meyers 1992). In part, reductions in the ratio of energy use per unit of economic activity were driven by a behavioral response to the energy price increases of the 1970s and early 1980s. More generally, however, the efficiency of energy use has been enhanced by a range of technologies related to buildings, transportation, and industrial production. Given current energy prices and the lack of interest in more stringent energy performance standards, progress towards enhanced energy efficiency has stalled in recent years. The Intergovernmental Panel on Climate Change (1996), however, estimates that worldwide energy use could be reduced by some 10–30% through the full implementation of cost-effective, energy-efficient technologies.

Opportunities to reduce NO_x emissions through reduced reliance on fossil fuels is a second possible control strategy. Of course, renewable energy sources such as biomass, animal power, and small-scale hydro have long histories in human societies and are still extensively used in developing countries. In the industrialized world, significant progress has been made in the use of solar and wind energy to provide significant flows of power at low environmental cost. These technologies, however, are characterized by high capital costs and reliability constraints that limit their general adoption (Holdren 1990). In a similar vein, nuclear power and large-scale hydroelectric dams – although free from the emissions of traditional air pollutants that are associated with fossil fuel combustion – face significant issues of cost, adaptability, and environmental impacts. These technologies are best suited to the generation of electric power, which constitutes a poor substitute for liquid fuels in transportation. Nuclear power raises questions of reactor safety, waste disposal, and weapons proliferation, while hydroelectric dams pose unique threats to natural environments and the people who populate them.

A broad range of technologies exists for reducing the amounts of NO_x generated by the combustion of specific fossil fuels. Because diatomic nitrogen is a chemically stable compound, its oxidation to NO_x involves high activation energies that are typically achieved only in regions of peak flame temperature. Since the relationship between flame temperature and emissions is strongly nonlinear, steps to reduce temperature are a major pathway to pollution control. In addition, NO_x formation is generally favored by the presence of excess oxygen in combustion gases. Carefully balancing the ratio of fuel to combustion air is thus also important in emissions control. Both combustion temperatures and fuel-air ratios are addressed through the use of flue gas recirculation and water injection in natural gas and oil boilers. These technologies achieve typical emissions reductions of 70–90% in practical applications (Seinfeld 1986). In coal-fired power plants the use of staged combustion methods is more common. This approach, in which fuel is burned in sequential stages in conditions of low oxygen availability and low combustion temperatures, reduces emissions by 30–50%. Emerging technologies involving advanced burner designs and the removal of NO_x from stack gases also have the potential to reduce energy-related NO_x emissions from stationary sources.

The technologies used to control NO_x emissions from vehicles are similar in principle but different in the details. In this context, pollution control technologies focus on achieving appropriate fuel-to-air ratios, exhaust gas recycling, and the removal of pollutants from exhaust gases using catalytic converters. Although technologies have improved substantially in the industrialized nations given the environmental regulations that emerged in the 1970s and 1980s, growth in transportation activity has offset these improvements so that NO_x emissions have been relatively stable. In addition, there is concern that catalytic converters actually increase emissions of nitrous oxide (N_2O), a greenhouse gas that generates a range of environmental harms (Dasch 1992). In developing countries that lack effective regulatory mechanisms, emissions of NO_x and other energy-related pollutants continue to pose serious threats to urban air quality and ecosystem health.

Policy approaches

Fertilizer N and livestock production

Given the characteristics of change in the N cycle discussed above, what policy approaches are available to mitigate or prevent environmental problems associated with fertilizer N use? In reviewing the policy options, three principles should be emphasized. First, although N cycling occurs at regional and global scales, policies are implemented and enforced at the national or provincial/state levels. Multinational efforts to control nitrogen loss to the environment are surely needed, but these efforts will require commitments from individual countries and the policy-makers within those countries.

Second, there is a wide range of direct and indirect policy instruments available to alter fertilizer N use at the national level, ranging from fertilizer subsidies, taxes, and regulations to exchange rate controls. Policies influencing the use and loss of fertilizer N are thus made by several institutions – often concurrently – including the Ministry (or Department) of Agriculture, the Central Bank, the Finance Ministry, and the Ministry of Environment and Natural Resources. These institutions may have opposing objectives, and communication between the groups is often poor or non-existent. Moreover, external advisory bodies, such as the World Bank, may play an important role in designing or endorsing policies related to fertilizer use.

Finally, officials implementing policies at the national level are usually different from the people engaged in international activities that deal with global N issues, such as the Framework Convention on Climate Change, the World Health Organization, and the World Trade Organization. Again, good communication between people within these international and national institutions is required to make global or regional initiatives on fertilizer N use and loss more effective.

While the following section is not a complete review, we will attempt to illustrate the policy dimensions of change in the nitrogen cycle by providing examples that have been employed at international, national and local/regional scales. At the national level, there are four sets of direct policy instruments that can be used to control N use and loss within the agriculture and livestock sectors: 1) subsidies and taxes on N fertilizer; 2) regulations on nitrate losses from agriculture and livestock activities; 3) investments in technology to increase N uptake by plants and reduce N loss (e.g., breeding efforts, nitrification inhibitors); and 4) expenditure on extension services and knowledge transfer to enhance the efficiency of N use through improved rates, timing, and placement of fertilizers. There are also several indirect policy measures that affect nitrogen use and loss within countries, which may be complementary to – or competitive with – direct policy instruments. Examples of such measures include commodity subsidies on crops or livestock products, land tenure policies, conservation and set-aside programs, credit subsidies, irrigation/water subsidies, macroeconomic and trade policies (and their impacts on exchange rates and interest rates), and marketing policies.

Taxes and subsidies on nitrogen fertilizer are the most direct and widespread means to influence N use in agriculture. With the introduction of Green Revolution seed technologies in the late-1960s and early-1970s, many developing countries subsidized fertilizers in order to encourage their use in conjunction with the new varieties. In other countries, fertilizers remained unsubsidized, or even implicitly taxed. During the past decade, price regimes for fertilizers have varied from free market pricing in Thailand to fully controlled pricing in Nigeria (Bumb & Baanante 1996). In countries where fertilizer supplies have not been constrained, fertilizer subsidies have promoted rapid growth in both fertilizer use and food production; examples include China, India, Indonesia, and Mexico.

In some of these cases, subsidies have become financial burdens on the economy and have led to overuse of fertilizer N (Conway 1997). In India, for example, fertilizer subsidies amounted to roughly 3% of the national budget in the early 1990s (Bumb & Baanante 1996). This subsidy rate is high even by OECD standards; total agricultural subsidies (inputs and outputs) in the OECD countries were $3.62 billion, or 1.3% of GDP, in 1998. The longer-term trend for subsidies, at least in the OECD countries, remains downward as governments increasingly introduce market-oriented farm policies. In 1986–1988, farm subsidies in the OECD countries averaged 2.1% of GDP (OECD 1999).

Taxes and subsidies for fertilizers are measured by the difference between international and domestic prices (accounting for transportation, insurance, and other marketing costs); if it is less expensive for a farmer to import fertilizer than to buy it domestically, the fertilizer is taxed, and vice versa for subsidies. Ratios of international to domestic fertilizer N prices have ranged from about 0.25 on the low side (tax) to about 1.5 on the high side (subsidy) in recent years. For example, in 1997, many African countries (e.g., Ethiopia, Kenya, and Zimbabwe) had ratios significantly below one, while several countries in Asia and South America (e.g., India and Argentina) had

Table 3. Ratio of fertilizer N to crop prices at the farm-gate.

Country	Ratio of N: wheat prices[1]	Ratio of N: corn prices[2]	Ratio of N: rice prices[3]
Ethiopia	2.40	7.79	–
Kenya	4.80	7.30	–
Zimbabwe	2.40	4.90	–
China	1.90	–	1.35
India	2.10	–	1.21
Pakistan	1.85	2.50	1.03
Japan	–	–	0.22
Argentina	5.80	2.40	–
Brazil	4.58	3.20	–
Mexico	3.30	1.30	–
World	2.00	2.50	1.16

[1] 1994, CIMMYT (1996); [2] 1997, CIMMYT (1999); [3] 1990, IRRI (1995).

ratios above one. The ratios are calculated from data collected by CIMMYT (1996), IRRI (1995), and the World Bank.

In many countries, ‘high’ (or ‘low’) fertilizer prices (as measured by historical prices or world prices) are compensated to some extent by high (or low) crop prices. Moreover, both sets of prices are often distorted by over- or under-valued exchange rates. An important policy variable for the objectives of enhanced food production and rural incomes is thus the ratio between N fertilizer prices and output prices, measured in the local currency at the farm gate. This ratio directly affects expected farm profits and can be used to gauge the potential farm-level impact of increasing fertilizer prices. For a given crop and year, this ratio varies widely among countries. Table 3 shows this variation for a subset of countries, commodities, and years.

The extent to which price policies should be used to force fertilizer producers and farmers to ‘internalize external costs’ of fertilizer N is debatable, and the answer depends importantly on the economic and policy context of the country. Given the essential role of nitrogen fertilizers in crop production in many regions of the world, increasing the costs to producers could lead to reduced yields and/or higher food prices for consumers. This outcome would be damaging for chronic food-deficient countries, such as those in Sub-Saharan Africa, whose fertilizer use is still less than one-fourth the world average. In these regions, increasing (rather than decreasing) nutrient use is a major objective of agricultural policy. In countries that are over-applying fertilizer N, one tractable policy is to reduce subsidies on fertilizers. If subsidies do not exist, alternative policy measures, such as regulations on the type, timing, and amount of fertilizer N applied, or taxes on imported fertilizer can be used to reduce N use and lower N losses.

In most developing countries, regulations on N use or loss are either non-existent or not enforced. Even in North America and Europe, monitoring and enforcement of non-point source pollution of nitrates policies is difficult, and there are few mandates on other forms of N loss. Both the U.S. and the European Union have regulations in place to limit N leaching losses from agricultural and livestock systems. These policies are compatible with the World Health Organization's recommendation that nitrate levels in drinking water should not exceed 50 milligrams of nitrate per liter of water (Bumb & Baanante 1996; Conway 1997). The Nitrate Directive of the European Union, formed in 1991, has as its major objective the control of net N supplies (supply minus uptake) to the soil beginning in 1999. Similarly in the U.S. the Clean Water Act was enacted in 1972 to improve national water quality. To renew impetus, the Clean Water Action Plan was developed jointly by nine Federal Agencies in 1998 to supplement the Clean Water Act. The Action Plan targets watershed protection as the main priority and provides communities with new resources to control pollution runoff and enhance natural resource stewardship (http://www.cleanwater.gov/action/). The Action Plan contains 111 key actions and was initiated with a 5-year funding proposal to provide approximately $2.3 billion in new funds. In 1999 the U.S. Congress funded $171 million of the $568 million requested for the Action Plan. The U.S. EPA estimates that 52% of the community water wells and 57% of the domestic water wells in the country contain nitrate (http://www.epa.gov/grtlakes/seahome/groundwater/src/overview.htm). In the U.S., individual states can implement their own N policy as long as it is compatible with national programs. The policies adopted by the state of Nebraska highlighted in Box 2 are an example.

Although regulations on N loss are not always effective, extension services and other forms of knowledge transfer have played an important role in helping farmers in North America and Europe to reduce N losses from agriculture. In these countries, fields are regularly tested, and farmers receive precise recommendations on application rates and timing, dependent on the soil type, irrigation practice, and previous crop. By contrast, N fertilizer applications often are not timed well to the plants' needs and flooded conditions in many developing countries. Many rice farmers in Asia, for instance, apply fertilizer directly to the water 1–3 weeks after transplanting, which results in large N losses to the atmosphere (Peoples et al. 1995). A critical policy need for the developing world is investment in and implementation of more

The Nebraska Department of Environmental Quality has been charged by new state legislation to require permitting of all livestock feeding operations larger than 1000 animal units. These permits require whole-farm nutrient management plans with easements for application of the expected manure load on neighboring farms if the amount of land owned by the livestock farm operation is not adequate to absorb the nutrient load in the manure. ('Adequate' is measured by a simple input-output formula taking into account manure production and the nutrient content of the manure, crop nutrient requirements for N, and an 'availability index' for the manure which relates to the rate at which the N it contains becomes available to the crop.)

Despite positive actions to reduce N loss from livestock, a more important problem is the loss of phosphorus, because the N/P ratio in manure is much lower than in plants. Thus, applying the manure at rates to satisfy crop N requirements leads to massive P accumulation in soil and a significant potential for P runoff to surface waterways. Eventually both the EPA and state regulators will need to focus more tightly on the P challenge of animal manure applications to farm land.

Box 2. Water quality policies adopted by the state of Nebraska, USA.

extensive training services on nitrogen use efficiency. Investments in fertilizer N management would complement further investments in plant breeding aimed at improving N uptake and allocation in crops.

Indirect policy measures may have an equal, if not larger, impact on N use efficiency and loss from agricultural systems. For example, macroeconomic policy that raises interest rates or reduces credit availability for farmers may lead to lower N application rates or greater N use efficiency. In many developing countries, fertilizer accounts for a large share of cash expenditure for small and medium farmers, and resource-poor farmers generally depend on borrowed funds to purchase fertilizer and other agricultural inputs. In Eastern Europe, the division of large state farms into small-holder units has led to a situation in which many small farmers do not have money or credit to purchase fertilizers; as a result, total fertilizer consumption in East Europe declined from 10 Tg of nutrients in 1986/87 to 3.7 Tg in 1996/97 (IFDC 1999).

A different type of land policy has influenced fertilizer rates in Western Europe. Land set-asides through the Common Agricultural Policy have contributed to an annual reduction in fertilizer consumption of 2.1% during the past ten years, with the largest reductions occurring in France, Germany, Italy, and the U.K. (IFDC 1999). The E.U. had an estimated set-aside rate of 10% for the 1998/99 harvest, double that of recent years. Although this program has dramatically reduced the area planted, it is not yet clear whether it has resulted in increased fertilizer use per hectare or increased fertilizer use efficiency (IFDC 1999).

It is clear from this discussion that no easy generalizations can be made about global nitrogen policy approaches. Each country has a different balance of policy objectives and budget constraints. National policies directly targeted toward nitrogen use and loss may also be confounded by other types of policies, including macroeconomic policies affecting exchange rates and interest rates. Some countries may choose to target nitrogen gases as part of a multi-gas approach towards reducing greenhouse gases within the framework of the Kyoto Protocol (Reilly et al. 1999); industrialized countries with already high rates of fertilizer N use and increasing N use efficiency would most likely enter into such an agreement. The agricultural policy of the Netherlands is an example and is highlighted in Box 3. For food deficit countries, particularly those which still underutilize fertilizer N, entering into an international agreement to control global N loss is improbable. The most interesting set of countries to watch with respect to international agreements on N loss include China and India, which are among the largest consumers of fertilizer N in the world. These countries have relatively strong agricultural sectors, but also escalating population- and income-driven food demands.

Energy sector

Efforts to reduce energy-related N emissions have been approached mainly through air pollution control policies. These policies were pioneered in the 1960s in response to perceived problems of urban air quality. Measures to improve the efficiency of energy use have also played a role (Box 4). Although it had long been recognized that emissions of particulates, SO_2, and NO_x imposed negative aesthetic impacts, the effects of these pollutants on human health were first recognized in the years following World War II. In 1948, an air pollution event occurred in the steel town of Donora, Pennsylvania that caused 6,000 illnesses and 20 fatalities in a population of 14,000 people (Snyder 1994). And in 1952, a smog event in the city of London caused 4,000 deaths over the course of a few days (Wise 1968). Public health officials came to recognize the cause-and-effect relationship between health risks and pollutant concentrations. Growing awareness of such risks led to calls for environmental regulation.

The United States Clean Air Act

In the United States, air pollution control policy is carried out under the auspices of the Clean Air Act (CAA), the major provisions of which date to 1970. The CAA requires the Environmental Protection Agency (EPA) to establish ambient air quality standards for so-called ‘criteria pollutants’ – particulates, SO_2, NO_x, tropospheric ozone, carbon monoxide, and lead – that

Dutch agriculture is among the most intensive in the world and typically has had high livestock stocking density and a high level of crop production per hectare. During the past 15 years the Dutch government has embarked on a policy to address the manure and ammonia problem (Policy 1999). The agricultural sector has invested heavily in efforts to implement environmentally friendly technology and management. In the Netherlands, agriculture contributes ∼30% of total phosphate and 75% of total N to surface waters. More than half of total acidification is traced to ammonia emissions, ∼90% of which are derived from agriculture. The main burden of phosphates and N from agriculture are from the over-application of livestock manure, the over-application of inorganic fertilizers and ammonia emissions.

In general, intensive livestock farms with large phosphate and mineral N and NH_3 losses constrained to a relatively limited land area, produce the greatest environmental risk. The standard unit for livestock density in the Netherlands is one dairy cow. From 1998 to 2002 a minerals accounting is obligatory only for farms with a livestock density of 2.5 livestock units (LU)/ha. In 2002 the standard will be lowered to 2 LU/ha. To limit N leaching, runoff, and NH_3 emissions, spreading manure in autumn and winter is banned. Use of techniques that minimize NH_3 emissions when manure is spread or stored is required.

Manure management policy: The Dutch animal manure policy is aimed at farms posing the highest environmental risks. A farm minerals accounting approach is used where an input-output book-keeping system relates total applications of fertilizers to production. If the losses, which are reported yearly, exceed the standards then a fine is imposed on the excess. Beginning in 1998, farms with more than 2.5 LU stocking rates are required to report mineral losses.

Ammonia policy: The Policy (1999) report indicates that of the ammonia losses to the atmosphere from Dutch agriculture ∼36% is derived from livestock housing and manure storage, ∼50% from manure spreading and ∼14% from waste deposition in the field from grazing animals. Such mitigation measures as low-emission application and covering manure storage tanks are suggested for implementation within the livestock production sector, which is indicated to account for 55% of agricultural NH_3 emissions. Other options proposed include selling less profitable animals, increasing milk yield per cow and improving animal feeds. Pig production is attributed a 30% share of agricultural NH_3 emissions and is decreased by the development of low-emission pig rearing units. The poultry sector accounts for ∼15% of total emissions and could be reduced by low-emission housing systems NH_3 policy. The NH_3 policy emphasis is on emission reduction. Farmers having stocking densities below 2 LU will deal only with basic mitigation measures while farmers having stocking densities over 2 LU will be obligated to construct low-emission housing. NH_3 policy targets a 70% reduction between 2000 and 2005 compared to 1980. Lowering emissions from manure application are expected to reduce emissions from slurry boom 50%, spreading harrows ∼60%, shallow injection ∼85% and deep injection ∼95%. Since small fines on N will likely not entice farmers to build low-emission housing, special requirements are made on housing systems to reduce NH_3 emissions.

Box 3. Policy on manure and ammonia emissions from agriculture in The Netherlands.

Since anthropogenic emissions of NO_X and other air pollutants are dominated by fossil fuel combustion, efforts to improve energy efficiency, measured in terms of the services obtained per unit of fuel consumption, have been a major focus of environmental policy. In the United States, the Corporate Average Fuel Economy (CAFE) standards, introduced in response to the 1970s oil shortage, require passenger cars to achieve an efficiency target of 27.5 miles per gallon (8.5 l/100 km), or 20.7 miles per gallon (11.35 l/100 km) for 'light trucks' (pickup trucks, minivans, and sport utility vehicles). It is widely recognized that these standards have led to substantial improvements in vehicle technology and reduced energy use. In a similar vein, the U.S. Appliance Efficiency Standards, which target the energy efficiency of refrigerators, washers, dryers, and other household equipment, are expected to save some 24 exajoules of energy from their initial adoption in 1990 through the year 2015 (Geller 1997).

The merits of policies to promote energy efficiency are not without controversy. Critics charge that the CAFE standards have forced consumers to purchase small vehicles that compromise performance while posing increased safety risks. Crandall and Graham (1989), for example, suggest that the vehicle size reductions induced by the CAFE standards may have caused as many as 3,900 fatalities for the 1989 model year alone. Greene (1998), in contrast, points to opportunities to enhance the safety of small cars through careful engineering. Although safer for their occupants, the large vehicles that are currently so popular impose heightened risks on third parties; a factor not considered by Crandall and Graham.

By way of comparison, the appliance efficiency standards provide evidence that well-designed policies can yield simultaneous energy and economic savings. These regulations, which are based explicitly on cost-effectiveness criteria, provide consumers with an estimated \$1.8 billion in net annual savings (Geller 1997). More generally, the Intergovernmental Panel on Climate Change (1996) concludes that the full adoption of least-cost energy-efficient technologies could reduce energy use per unit of economic activity by some 10–30% in the industrial economies of Europe and North America.

Although energy efficiency can clearly contribute to the achievement of environmental policy goals, the links between reduced energy use and NO_X emissions are not entirely clear-cut. In the United States, the Clean Air Act requires passenger vehicles to meet uniform technology standards that specify maximum levels of pollutant emissions per distance traveled. Since all passenger cars are held to the same standards, NO_X emissions are partially decoupled from energy efficiency. Until recently, pickup trucks, minivans, and sport utility vehicles faced less stringent requirements than automobiles with respect to both fuel economy and emissions standards. In December of 1999, the Clinton administration issued new regulations under which these 'light trucks' will be held to the same emissions standards as automobiles. The disparity in fuel economy requirements remains in place, however, providing incentives for the production and sale of large, energy-intensive vehicles.

Box 4. Energy efficiency and NO_X.

are known to have serious health impacts on affected populations. The statute requires EPA to set standards at the level required 'to protect public health' based on the best available experimental and epidemiological data.

Air quality standards for criteria pollutants set national targets that are implemented and enforced through a two-part policy approach. First, the CAA requires state governments, in consultation with EPA, to construct so-called 'state implementation plans' (SIPs) to achieve air quality standards within the state's boundaries. A SIP imposes specific emissions standards on major stationary sources of pollution such as industrial facilities and power plants. State administrators are granted considerable flexibility regarding the criteria used to distribute required emissions reductions between sources provided that air quality achieves federal standards. Second, the federal government itself defines and enforces national emissions standards for both new stationary sources of pollution ('New Source Performance Standards') and transportation equipment. Under these rules, new power plants and industrial facilities must meet stringent, technology-based standards that set upper bounds on allowable emissions of criteria pollutants. In a similar vein, passenger automobiles must be equipped with catalytic converters and related pollution abatement measures.

The ambient air quality standard for NO_x is currently set at 0.053 ppm (USEPA 1999). This standard is reviewed at 5-y intervals to account for advances in scientific knowledge. In addition, NO_x emissions are regulated with an eye towards reducing ambient concentrations of tropospheric ozone. NO_x is a well-known ozone precursor that, in conjunction with hydrocarbon emissions, gives rise to photochemical smog. The impacts of the CAA on NO_x emissions in the United States have been quite substantial. Although emissions were rising rapidly prior to 1970 with the growth of energy use, transportation activity, and industrial production, U.S. NO_x emissions in 1995 were limited to 21.8 Tg – an increase of only 11% between 1970 and 1997 despite substantial growth in population and economic activity (USEPA 1998b). Given the success of State Implementation Plans, New Source Performance Standards, and emissions standards for new vehicles, only 13 U.S. cities violated air quality standards on ten or more days in 1997 (USEPA 1998c).

Despite such success, environmentalists have called for more aggressive policies to address the ecological impacts of energy-related NO_x emissions. Given the CAA's traditional focus on improving urban air quality, questions of long-range pollution transport – in which NO_x is involved as a precursor of both tropospheric ozone and acid deposition – have been difficult to address under the prevailing regulatory framework. On the one hand, state regulators have traditionally had no reason to consider the impacts of their own state's emissions on air quality beyond their own jurisdictions. On the other hand, a mechanism for achieving coordinated emissions reductions across state boundaries was, until recently, not embodied in the statute.

This problem was partially addressed by the 1990 Clean Air Act amendments, which aimed in part to reduce the ecological impacts of acid deposition caused by the long-range transport of SO_2 and NO_x. The amendments call for a 50% reduction in sulfur emissions from electrical generating stations in the Midwest and Northeast, along with smaller reductions in NO_x emissions from these same sources. The differential treatment of sulfur and nitrogen was based in part on the perception that SO_2 is ecologically more damaging than NO_x. In part, this approach was based on the belief that emissions abatement costs would be lower for sulfur than for nitrogen since SO_2 emissions could be achieved through a mere shift from high- to low-sulfur fuels. In September of 1998, addition, EPA promulgated a new regulation known as the 'NO_x SIP Call' to address interstate transboundary flows of nitrogen oxides in order to reduce tropospheric ozone pollution. Under this rule, 22 states in the Northeastern U.S. must reduce their NO_x emissions by an average of 28% during the summer months, the period for which ambient ozone concentrations are likely to exceed federal standards (USEPA 1998a).

The convention on long-range transboundary air pollution

While the Clean Air Act of 1970 began to address the human health impacts of U.S. urban air pollution, evidence of a more complex problem was turning up in Europe, where studies were linking forest dieback and the acidification of Scandinavian lakes to continental emissions of SO_2 and NO_x. When scientists established that air pollutants could be transported hundreds or even thousands of kilometers across national frontiers, it became clear that effective response strategies would require international cooperation. In 1972, the participants of the United Nations Conference on the Human Environment in Stockholm declared that nations must ensure that activities carried out in their territories do not result in environmental damage in other states. Consequently, a 1979 ministerial meeting of the U.N. Economic Commission for Europe (UNECE) was convened in Geneva to address the region's environmental problems. The result was the Convention on Long-Range Transboundary Air Pollution (LRTAP), an agreement that was signed by all 35 (now 44) member countries.

The goal of LRTAP is to 'limit and, as far as possible, gradually reduce and prevent air pollution [using] the best available technology which is economically feasible.' Establishing the convention-protocol model that has since been used in international agreements on ozone depletion and climate change, the agreement led to a series of protocols mandating targets and timetables for pollution abatement as well as the long-term funding of the European Monitoring and Evaluation Program (EMEP) in 1984. EMEP is a main source of data both on air pollution levels and transboundary fluxes, gathering data at 100 monitoring stations in 25 countries (see http://projects.dnmi.no/˜emep/index.html). The first protocol was signed in Helsinki in 1985, where signatories agreed to reduce SO_2 emissions by at least 30% relative to 1980 levels. Although neither the United States nor the

United Kingdom signed this agreement, overall sulfur emissions in UNECE countries had been reduced by over 50% through 1993 (UNECE 2000a).

A series of subsequent protocols has addressed the deleterious impacts of NO_x (1988), volatile organic compounds (1991), heavy metals (1998), and persistent organic pollutants (1998). A second sulfur protocol (1994) stiffened the requirements of the Helsinki accord. The most recent agreement – the Protocol to Abate Acidification, Eutrophication and Ground-level Ozone (1999) – deals with multiple effects and multiple pollutants, with a special emphasis on SO_2, NO, volatile organic compounds, and NH_3. Both the NO_x and the 'Multiple Effects' Protocols deal with human effects on the nitrogen cycle.

With 25 original signatories and 27 ratifications, the Sofia Protocol Concerning the Control of Emissions of Nitrogen Oxides or their Transboundary Fluxes entered into force in 1991. This agreement obliges its parties to freeze NO_x emissions at 1987 (or, for the United States, 1978) levels by 1994. The agreement prescribes both technical means for reducing NO_x emissions using specific technologies and policy measures that include the establishment of national emissions standards for all new and existing sources of NO_x. As of 1994, most signatory nations had met or exceeded their abatement targets, with a total emissions reduction of 9% (UNECE 2000a).

Article 6 of the NO_x Protocol laid the foundation for future emissions reductions based on the so-called 'Critical Loads Approach.' A critical load is "a quantitative estimate of the exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge." The article aims to ensure that critical loads of nitrogen are not exceeded due to anthropogenic emissions, assigning the tasks of evaluation and monitoring to individual nations. This framework set the stage for the Multiple Effects Protocol that was signed in Gothenburg in November of 1999.

The Multiple Effects Protocol sets differentiated emissions ceilings for its 29 parties based on the severity of the effects to which each contributes (both within its borders and abroad) and the cost-effectiveness of emissions abatement (UNECE 2000b). This approach builds on research carried out by the EMEP in the 1980s and 1990s. Through 2010, UNECE (2000b) estimates that this agreement will yield emissions reductions of 63% for SO_2, 41% for NO_x, 40% for volatile organic compounds, and 17% for NH_3 relative to 1990 levels. For NO_x, the policy approach taken by the Multiple Effects Protocol largely resembles the earlier Sofia Protocol, setting overall emissions ceilings and limiting pollution from both new and existing sources. Unlike the Sofia agreement, however, the Multiple Effects Protocol allows countries to choose their own pollution control strategies as long as they attain the stipulated emissions targets.

In addition to NO_x, the Multiple Effects Protocol establishes measures to control NH_3 emissions, directing parties to publish and disseminate guides to good agricultural practices to abate emissions within their borders. These guidelines must address nitrogen management (taking the full nitrogen cycle into account), livestock feeding strategies, manure spreading techniques, manure storage, animal housing, and the prospects for limiting ammonia emissions from the use of mineral fertilizers.

According to UNECE (2000b), the full implementation of the Multiple Effects Protocol would: (a) reduce the land area affected by excess acidification from 93 to 15 million hectares; (b) reduce eutrophication by 35%; (c) cut the number of lives lost from exposure to ozone and particulates by 47,500 per annum; and (d) lower the exposure of vegetation to high ozone levels by 44%. The protocol will enter into force 90 days after it is ratified by at least 16 of its 29 signatories. At present, the agreement has a large and growing number of parties, and there is cause for optimism regarding its implementation.

Overall, LRTAP and its protocols have been quite effective in reducing air pollution. In addition, these agreements have fostered a large body of scientific research and established a framework for international cooperation.

Summary

The human induced input of reactive N into the global biosphere has increased to approximately 150 Tg N each year (Figure 1) and is expected to continue to increase for the foreseeable future. The need to feed and to provide energy for the growing world population drives this trend. This increase in reactive N comes at, in some instances, significant monetary and nonmonetary costs to society through increased emissions of NO_x, NH_3, N_2O and NO_3^- and deposition of NO_y and NHx.

In the atmosphere, increases in tropospheric ozone and acid deposition (NO_y and NHx) have lead to acidification of aquatic and soil systems and in reductions in forest and crop system production. Changes in aquatic systems as a result of nitrate leaching have led to decreased drinking water quality, eutrophication, hypoxia and decreases in aquatic plant diversity, for example.

On the other hand, increased deposition of biologically available N may have increased forest biomass production and may have contributed to increased storage of atmospheric CO_2 in plant and soils. Most importantly, synthetic production of fertilizer N has contributed greatly to the remarkable increase in food production that has taken place during the past 50 years.

In 1994 fuel combustion fixed ~21 Tg of N while food production fixed approximately 120 Tg of N. There are many possibilities for limiting reactive N emissions from fuel combustion, and in fact, great strides have been made during the past decades. The control and reduction of the introduction of new reactive N and in curtailing the movement of this N in food production is even more difficult. The particular problem comes from the fact that most of the N that is introduced into the global food production system is not converted into usable product, but rather reenters the biosphere as a surplus (Table 1). Policy issue problems are further exacerbated by the fact that most reactive N species are highly mobile through atmospheric transport or flow through aquatic systems. Reactive N produced in one region or country may cross territorial boundaries. Policies for control are typically found on a state or national basis. There are examples, such as the U.S. Clean Air Act of 1970 and later amendments, that have helped minimize the increase in NO_x emissions from combustion. The U.S. NO_x emissions have increased by ~6% while human population has increased by ~30% during the past 30 years.

The difficulties for policy development to control unwanted reactive N release are confounded by the fact that much of the reactive N release is involved with food and energy production. Global policy on N in agriculture is further difficult because many countries need to increase the input of fertilizer N to increase food production to raise nutritional levels or to keep up with population growth. There is a wide range of direct and indirect policy instruments available to alter fertilizer N use at the national level, ranging from fertilizer subsidies, taxes, and regulations to exchange rate controls. Policies influencing the use and loss of fertilizer N are thus made by several institutions – often concurrently – including the Ministry (or Department) of Agriculture, the Central Bank, the Finance Ministry, and the Ministry of Environment and Natural Resources. These institutions may have opposing objectives, and communication between the groups is often poor or non-existent. Moreover, external advisory bodies, such as the World Bank, may play an important role in designing or endorsing policies related to fertilizer use.

Although N cycling occurs at regional and global scales, policies are implemented and enforced at the national or provincial/state levels. Multinational efforts to control nitrogen loss to the environment are surely needed, but these efforts will require commitments from individual countries and the policy-makers within those countries. Officials implementing policies at the national level are usually different than the people engaged in international activities that deal with global N issues, such as the Framework Convention on Climate Change, the World Health Organization, and the World Trade Organization. Again, good communication between people within these international and national institutions is required to make global or regional initiatives on fertilizer N use and loss and fossil fuel consumption more effective.

Acknowledgments

This work was initiated as part of the International SCOPE N Project, which received support from both the Mellon Foundation and from the National Center for Ecological Analysis and Synthesis, a Center funded by NSF (Grant #DEB-94-21535), the University of California at Santa Barbara and the State of California.

References

Aber JD, Magill A, McNulty SG, Boone RD, Nadlehoffer KJ, Downs M & Hallett R (1995) Forest biogeochemistry and primary production altered by nitrogen saturation. Water, Air and Soil Poll. 85: 1665–1670

Aber J, McDowell W, Nadelhoffer K, Magill A, Berntson G, Kamakea M, McNulty S, Currie W, Rustad L, & Fernandez I (1998) Nitrogen saturation in temperate forest ecosytems. Bioscience 48: 921–934

Abrahamsen G & Stuanes AO (1998) Retention and leaching of N in Norwegian coniferous forests. Nut. Cycling in Agroecosys. 52: 171–178

Almanac for Agriculture of China (1982, 1997) (in Chinese)

Ayers RU, Schlesinger WH & Socolow RH (1994) Human impacts on the carbon and nitrogen cycles. In: Socolow RH, Andrews C, Berkhout R & Thomas V (Eds) Industrial Ecology and Global Change (pp 121–155). Cambridge University Press, New York, New York, USA

Badiane O & Delgado CL (1995) A 2020 Vision for Food, Agriculture, and the Environment in Sub-Saharan Africa. International Food Policy Research Institute, Washington D.C. 56 pp

Benkovitz CM, Scholtz MT, Pacyna J Tarras n L, Dignon J, Voldner EC, Spiro PA, Logan JA & Graedel TE (1996) Global gridded inventories of anthropogenic emissions of sulfur and nitrogen. J. of Geophys. Res. 101: 29,239–29,253

Bergesen HE & Parmann G (1997) Green Globe Yearbook. Oxford University Press, New York.

Bleken MA (1997) Food consumption and nitrogen losses from agriculture. In: Lag J (Ed.) Some Geomedical Consequences of Nitrogen Circulation Processes. Proceedings of an International Symposium, 12–13 June, 1997 (pp 19–31). The Norwegian Academy of Science and Letters, Oslo, Norway

Bleken MA & Bakken LR (1997) The nitrogen cost of food production: Norwegian Society. Ambio 26: 134–142

Bouwman AF (1997) Long-term scenarios of livestock-crop-land use interactions in developing countries. FAO Land and Water Bulletin 6. FAO Rome. 145 p

Bouwmann AF & Booij H (1998) Global use and trade of feedstuffs and consequences for the nitrogen cycle. Nutrient Cycling in Agroecosystems 52: 261–267

Broadbent FE & Carlton AB (1978) Field trials with isotopically labeled nitrogen fertilizer. In: Nielsen DR & MacDonald JG (Eds) Nitrogen in the Environment (pp 1–43). Academic Press, Inc. New York

Brown LR (1999) Feeding nine billion. In: Brown LR, Flavin C & French H (Eds) State of the World 1999. A World Watch Institute Report on Progress Toward a Sustainable Society (pp 115–132). W. Norton & Company, NY

Bumb BL & Baanante CA (1996) The role of fertilizer in sustaining food security and protecting the environment to 2020. Food, Agriculture, and the Environment Discussion Paper 17. International Food Policy Research Institute. Washington, D.C. USA. 54 pp

CIMMYT (1996) World Wheat Facts and Trends 1995/96. D. F., CIMMYT, Mexico

CIMMYT (1999) World Maize Facts and Trends 1997/98. D. F., CIMMYT, Mexico

Cole CV, Cerri C, Minami K, Mosier A, Rosenberg N & Sauerbeck D (1996) Agricultural options for mitigation of greenhouse gas emissions. In: Watson RT, Zinyowera MC & Moss RH Moss (Eds). Climate Change 1995. Impacts, Adaptations and Mitigation of Climate Change: Scientific Technical Analyses. Published for the Intergovernmental Panel on Climate Change, Chapter 23 (pp 745–771). Cambridge University Press, Cambridge, UK

Conway G (1997) The Doubly Green Revolution: Food for All in the 21st Century. Cornell University Press

Crandall, RW & Graham JD (1989) The effect of fuel economy standards on automobile safety. Journal of Law and Economics 32: 97–118

Cui YT, Cheng X, Han CR & Li RG (1998) Nitrogen utilization efficiency of rice and nitrogen leaching in Taihu Lake watershed of south Jiangsu. J. of China Agric. Univ. 3(5): 51–54 (in Chinese)

Dasch, JM (1992) Nitrous-oxide emissions from vehicles. Journal of the Air and Waste Management Association 42: 63–67

De Jager D, Blok K & van Brummelen M (1998) Cost-Effectiveness of Emission-Reducing Measures of Nitrous Oxide in The Netherlands. Ecosystem Report M704. Utrecht, The Netherlands

Delmas R, Serca D, & Jambert C (1997) Global inventory of NO_X sources. Nut. Cycling in Agroecosys. 48: 51–60

Downing JA, Baker JL, Diaz RJ, Prato T, Rabalais NN & Zimmerman RJ (1999) Gulf of Mexico Hypoxia: Land and Sea Interactions. Council for Agricultural Science and Technology, Task Force Report No 134. Ames, Iowa. 44 p

Ellis EC & Wang SM (1997) Sustainable traditional agriculture in the Tai Lake Region of China. Agriculture Ecosystems and Environment 61: 177–193

Ellis EC, Li RG, Yang LZ & Cheng X (2000a) Long-term change in village scale ecosystems in China using landscape and statistical methods. Ecological Applications 10: 1057–1073

Ellis EC, Li RG, Yang LZ & Cheng X (2000b) Changes in village-scale nitrogen storage in China's Tai Lake Region. Ecological Applications 10: 1074–1089

FAO, United Nations Food and Agricultural Organization (1999) FAOSTAT: Agricultural Data, are available on the world wide web: (http://www.apps.fao.org/cgi-bin/nph-db.pl?subset=agriculture)

Ferm M (1998) Atmospheric ammonia and ammonium transport in europe and critical loads – A review. Nutrient Cycling in Agroecosystems 51: 5–17

Fertilizers and Agriculture (January 1998) Is the Global Nitrogen Cycle Out of Balance? International Fertilizer Industry Association, Paris. 1 p

Galloway J N, Levy H II & Kasibhatla PS (1994) Year 2020: Consequences of population growth and development on the decomposition of oxidized nitrogen. Ambio 23: 120–12

Galloway JN, Schlesinger WH, Levy H II, Michaels A & Schnoor JL (1995) Nitrogen fixation: Anthropogenic enhancement-environmental response. Global Biogeochem. Cyc. 9: 235–252

Geller H (1997) National appliance efficiency standards in the USA: Cost-effective Federal regulations. Energy and Buildings 26(1): 101–109

Greene DL (1998) Why CAFE worked. Energy Policy 26: 595–613

Hendriks CA, de Jager D & Blok K (1998) Emission Reduction Potential and Costs for Methane and Nitrous Oxide in the EU-15. Interim Report. Ecofys report M714. Utrecht, The Netherlands

Holdren JP (1990) Energy in transition. Sci. Am. 263: 157–163

Holland EA & Lamarque JF (1997) Modeling bio-atmospheric coupling of the nitrogen cycle through NO_x emissions and NO_y deposition. Nutrient Cycling in Agroecosys. 48: 7–24

Howarth RW, Billen G, Swaney D, Toronsend A, Joworski N, Lajtha Downing JA, Elmgren R, Caraco N, Jordan T, Berendse F, Freney J, Kudeyarov V, Murdoch P & Zhao-liang Z (1996) Regional nitrogen budgets and riverine N and P fluxes for the drainages to the North Atlantic Ocean: Natural and human influences. Biogeochem. 35: 181–226

Hutchinson GL & Viets FG (1969) Nitrogen enrichment of surface water by absorption of ammonia volatilized from cattle feedlots. Sci. 166: 514–515

IFDC (1999) International Fertilizer Development Center. World Fertilizer Supply/Demand Situation. IFDC, Muscle Shoals, Alabama, February 1999

IPCC (1996) Intergovernmental Panel on Climate Change. Climate Change 1995: Economic and Social Dimensions of Climate Change. Cambridge University Press, New York

IPCC (1995) Intergovernmental panel on climate change. In: Houghton JT (Ed.) Climate Change 1994. Radiative Forcing of Climate Change and an Evaluation of the IPCC IS92 Emission Scenarios. Published for the IPCC, Cambridge University Press, Cambridge UK. 337 pp

IPCC (1997) Intergovernmental panel on climate change. In: Houghton JT, Meira Filho LG, Lim B, Trennton K, Mamaty I, Bonduki Y, Griggs DJ & Callander BA (Eds) Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories, Vol. 1–3

IRRI (1995) International Rice Research Institute. World Rice Statistics, Los Banos, The Philippines

Lal R, Kimble JM, Follett RF & Cole CV (1998) The Potential of U.S./Cropland to Sequester Carbon and Mitigate the Greenhouse Effect. Ann Arbor Press, Chelsa, MI. 128 p.

Legg JO & Meisinger JJ (1982) Soil nitrogen. In: Stevenson FJ (Ed.) Nitrogen in Agricultural Soils. Agronomy Monograph No. 22 (pp 503–566). ASA-CSSA-SSSA, Madison, WI

Leuck D, Haley S, Liapis P & McDonald B (1995) The EU Nitrate Directive and CAP Reform: Effects on Agricultural Production, Trade, and Residual Soil Nitrogen. Report 255. Economic Research Service. U.S. Department of Agriculture, Washington, DC

Li QK, Zhu ZL & Yu TR (1998) Fertilizers and manures in the sustainable agriculture of China. In: Jiangxi Science and Technology Publishers (pp 120–129). (in Chinese) Nanchang

Ma LS (1997) Nitrogen management and environmental and crop quality. In: Zhu ZL, Wen ZX & Freney JR (Eds) Nitrogen in Soils of China, Developments in Plant and Soil Sciences 74 (pp 303–321) Kluwer Academic Publishers, Dordrecht, The Netherlands

Matson PA, Naylor R & Ortiz-Monasterio I (1998) Integration of environmental, agronomic, and economic aspects of fertilizer management. Science 280: 112–115

Matson PA, Parton WJ, Power AG & Swift MJ (1997) Agricultural intensification and ecosystem properties. Science 277: 504–508

McTaggart IP, Clayton H, Parker J, Swan L & Smity KA (1997) Nitrous oxide emissions from grassland and spring barley, following N fertiliser application with and without nitrification inhibitors. Biology and Fertility of Soils 25: 261–268

Minami K (1994) Effect of nitrification inhibitors and slow-release fertilizer on emission of nitrous oxide from fertilized soils. In: Minami K, Mosier A & Sass R (Eds) CH_4 and N_2O-Global Emissions and Controls from Rice Fields and Other Agricultural and Industrial Sources (pp 187–196). NIAES, Yokendo, Tokyo

Mosier AR, Duxbury JM, Freney JR, Heinemeyer O & Minami K (1998) Assessing and mitigating N_2O emissions from agricultural soils. Climatic Change 40: 7–38

Müller JF (1992) Geographical distribution and seasonal variation of surface emissions and deposition velocities of atmospheric trace gases. J. Geophys. Res. 97: 3787–3804

NRC (National Research Council) (1993) Soil and Water Quality, An Agenda for Agriculture, Committee on Long-Range Soil and Water Conservation, Board on Agriculture, National Research Council, National Academy Press, Washington, DC. 516 p

OECD (1999) Orginazation of Economic Cooperation and Development. Data reported in Economist, "Farm Subsidies", June 5

Olsthoorn CSM & Fong NPK (1998) The anthropogenic nitrogen cycle in the Netherlands. Nut. Cycling in Agroecosys. 52: 269–276

Organization for Economic Cooperation and Development (1997) OECD Environmental Data Compendium. OECD, Paris

Peoples MB, Mosier AR & Freney JR (1995) Minimizing gaseous losses of nitrogen. In: Bacon PE (Ed.) Nitrogen Fertilization in the Environment (pp 565–602). Marcel Dekker, Inc

Policy (1999) Policy Document on Manure and Ammonia, The Netherlands. HYPERLINK http://www.minlnv.nl/international/policy/green/notutmm.html

Reilly J, Prinn RG, Harnisch J, Fitzmaurice J, Jacoby HD, Kicklighter D, Stone PH, Sokolov AP & Wang C (1999) Multi-Gas Assessment of the Kyoto Protocol. MIT Joint Program on the Science and Policy of Global Change. Report No. 45. 14 p

Schipper L & Meyers S (1992) Energy Efficiency and Human Activity. Cambridge University Press, New York

Seinfeld JH (1986) Atmospheric Chemistry and Physics of Air Pollution. John Wiley and Sons, New York

Shoji S & Kanno H (1994) Use of polyolefin-coated fertiizers for increasing fertilizer efficiency and reducing nitrte leaching and nitrous oxide emissions. Fertilizer Research 39: 147–152

Smil V (1990) Nitrogen and phosphorus. In: Turner BL II, Clark WC, Kates RW, Richards JF, Mathews JT & Meyer WB (Eds) The Earth as Transformed by Human action (pp 423–436). Cambridge University Press, Cambridge, England

Smil V (1991) Population growth and nitrogen: An exploration of a critical existential link. Population and Development Review 17: 569–601

Smil V (1999) Nitrogen in crop production: An account of global flows. Global Biogeochem. Cyc. 13: 647–662

Snyder LP (1994) The death-dealing smog over Donora, Pennsylvania: Industrial air pollution, public health policy, and the politics of expertise. Environmental History Review 18: 117

United Nations Economic Commission for Europe (UNECE) (2000a) Convention on Long-Range Transboundary Air Pollution. Environment and Human Settlements Division, Geneva. http://www.unece.org/env/lrtap

United Nations Economic Commission for Europe (UNECE) (2000b) Protocol to Abate Acidification, Eutrophication and Ground-Level Ozone. Environment and Human Settlements Division, Geneva. http://www.unece.org/env/lrtap/pdfbrochure/PolluEng_intro.pdf

USDA (United States Department of Agriculture) (1997) Agricultural Statistics 1997. United States Government Printing Office, Washington DC

UNEP/UNIDO (United Nations Environment Programme/United Nations Industrial Development Organization) (1996) Mineral Fertilizer Production and the Environment. Technical Report No. 26. In collaboration with the International Fertilizer Industry Assoc., Paris. 150 pp

U.S. Environmental Protection Agency (1998a) Fact Sheet: Final Rule for Reducing Regional Transport of Ground-Level Ozone. Office of Air and Radiation, Washington DC

U.S. Environmental Protection Agency (1998b) National Air Pollutant Emissions Update, 1900–1997. Office of Air Quality Planning and Standards, Research Triangle Park, North Carolina

U.S. Environmental Protection Agency (1998c) National Air Quality and Emissions Trend Report, 1997. Office of Air Quality Planning and Standards, Research Triangle Park, North Carolina

U.S. Environmental Protection Agency (1999) National Ambient Air Quality Standards. Office of Air and Radiation, Washington, DC

Van der Hoek KW (1998) Nitrogen efficiency in global animal production. Environmental Pollution 102: 127–132

Vitousek PM & Matson PA (1993) Agriculture, the global nitrogen cycle, and trace gas flux. In: Oremland RS (Ed.) The Biogeochemistry of Global Change: Radiative Trace Gases (pp 193–208). Chapman & Hall, New Youk, New York, USA

Vitousek PM, Aber J, Howarth RW, Likens GE, Matson PA, Schindler DW, Schlesinger WH & Tilman DG (1997) Human alteration of the global nitrogen cycle: Causes and Consequences. Issues in Ecology 1: 1–15

Wise W (1968) The World's Worst Air Pollution Disaster. Rand McNally, Chicago

Yan WJ, Yin CQ & Yu SM (1998) Nutrient budgets and biogeochemistry in an experimental agricultural watershed in Southeastern China. Biogeochem. (in press)

Zhu ZL (1997) The fate and management of chemical fertilizer nitrogen in agro-ecosystems. In: Zhu ZL, Wen QX & Freney JR (Eds) Nitrogen in Soils of China, Chap. 11. Kluwer Academic Publishers, Dordrecht/Boston/London

Zhu ZL (1998a) Fertilizers in relation to agriculture and environment. Exploration of Nature 17(4) 25–28 (in Chinese)

Zhu ZL (1998b) The present situation and problems of nitrogen fertilizer use and the strategies. In: Li QK, Zhu ZL & Yu TR (Eds) Fertilizers and Manures in the Sustainable Agriculture of China (pp 38–51). Jiangxi Science and Technology Publishers (in Chinese). Nanchang

Zhuang WL, Tian Z, Zhang N & Li K (1995) Investigation of nitrate pollution in ground water due to nitrogen fertilization in agriculture in north China. Plant Nutrition and Fertilizer Sciences 1: 80–87 (in Chinese)