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# MEETING CEREAL DEMAND WHILE PROTECTING NATURAL RESOURCES AND IMPROVING ENVIRONMENTAL QUALITY

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■ **Abstract** Agriculture is a resource-intensive enterprise. The manner in which food production systems utilize resources has a large influence on environmental quality. To evaluate prospects for conserving natural resources while meeting increased demand for cereals, we interpret recent trends and future trajectories in crop yields, land and nitrogen fertilizer use, carbon sequestration, and greenhouse gas emissions to identify key issues and challenges. Based on this assessment, we conclude that avoiding expansion of cultivation into natural ecosystems, increased nitrogen use efficiency, and improved soil quality are pivotal components of a sustainable agriculture that meets human needs and protects natural resources. To achieve this outcome will depend on raising the yield potential and closing existing yield gaps of the major cereal crops to avoid yield stagnation in some of the world's most productive systems. Recent trends suggest, however, that increasing crop yield potential is a formidable scientific challenge that has proven to be an elusive goal.

CONTENTS

INTRODUCTION . . . 316
ARABLE LAND RESOURCES . . . 318
  Estimating Land Reserves . . . 318
  Case Studies: Sub-Saharan Africa and China . . . 319
  Preserving Natural Ecosystems . . . 320
YIELD POTENTIAL AND EXPLOITABLE YIELD GAPS . . . 321
  Definitions . . . 321
  Importance of Maintaining an Exploitable Yield Gap . . . 321
  Estimating Trends in Rice Yield Potential . . . 324
  Estimating Trends in Yield Potential of Maize and Wheat . . . 327
  Are Existing Yield Gaps Large Enough? . . . 328

NITROGEN EFFICIENCY AND TRENDS IN NITROGEN FERTILIZER USE . . . 330
Inorganic Versus Organic Nitrogen Sources . . . 330
Nitrogen Efficiency at the Field Level . . . 331
Global Trends in Cereal Production and Nitrogen Fertilizer Use . . . 334
Disaggregating Trends in Cereal Yields and Nitrogen Fertilizer Use . . . 336
Projection of Future Nitrogen Fertilizer Requirements . . . 340
Improving Nitrogen Use Efficiency . . . 342
CARBON SEQUESTRATION, GREENHOUSE FORCING, AND SOIL QUALITY . . . 343
Carbon Sequestration and Greenhouse Gas Emissions . . . 343
Soil Quality, Nitrogen Requirements, and Greenhouse Gas Emissions . . . 346
CONCLUSIONS . . . 349

# INTRODUCTION

Agriculture currently appropriates a substantial portion of the Earth's natural resources. Crop production, pasture, and livestock grazing systems occupy 38% of total land area (1). Water used for irrigation accounts for 80% of all freshwater consumption (2). Nitrogen (N) fertilizer applied to agricultural land comprises more than 50% of the global reactive[1] N load attributable to human activities (3). The use of these resources has a number of negative environmental consequences (4–6). Land conversion from natural forests, wetlands, and grasslands to highly productive but simplified agroecosystems results in a substantial reduction in biodiversity on the converted land and a decrease in habitat for displaced wildlife and plant communities. Irrigation withdrawals from river systems and water bodies alter riparian habitat and reduce water quality necessary to support wildlife and native plant populations. Nitrogen losses associated with use of N fertilizer can result in nitrate contamination of water resources and increased emissions of nitrous oxide ($N_2O$), a potent greenhouse gas with a forcing potential about 300-fold greater than $CO_2$. Reduced water quality from irrigation withdrawals and nutrient losses from agricultural runoff have a negative impact on aquatic recreational activities that depend on pristine rivers, lakes, and coastlines.

Although production trends of the past 40 years have kept pace with food demand (Figure 1), at issue is whether the projected increases in food requirements can be met while protecting natural resources for future generations. Grain demand is expected to increase at a faster rate than population growth because economic development and urbanization will result in greater per capita consumption of livestock products in developing countries, where more than 95% of the population

[1]Reactive N refers to all N compounds in the atmosphere and biosphere that are biologically, photochemically, or radiatively active. The reactive N pool includes inorganic reduced (e.g., $NH_3$, $NH_4^+$) and oxidized (e.g., $NO_x$, $HNO_3$, $N_2O$, $NO_3^-$) compounds and organic compounds (e.g., urea, amines, proteins, amides).

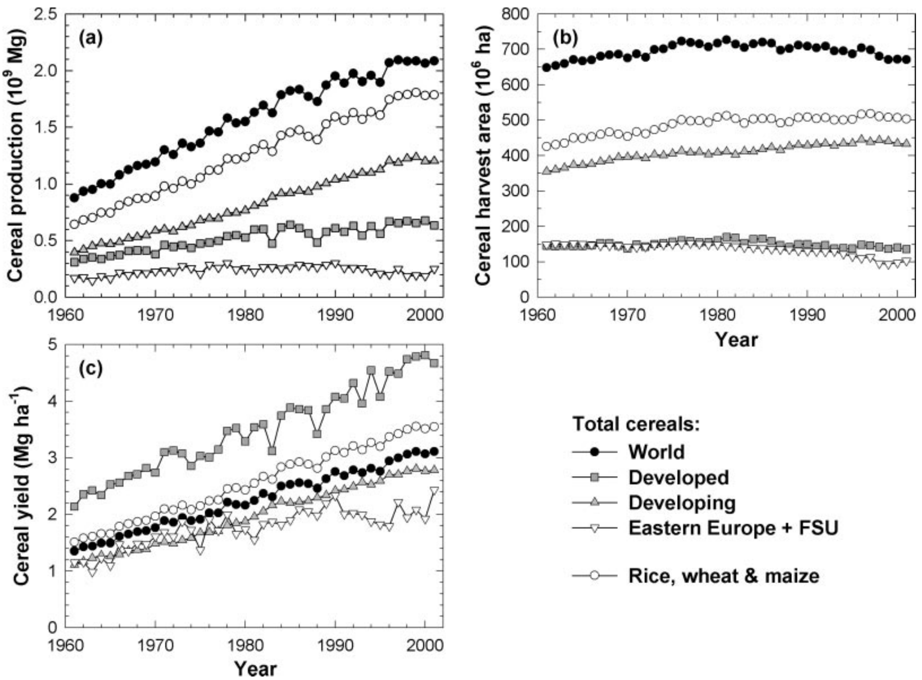



**Figure 1** Trends in global cereal production, harvested area, and yield from 1960 to 2001. Developed countries include those in Western Europe and North America, Australia, New Zealand, South Africa, Israel, and Japan. Developing countries include those in Latin America, Africa, Near East, Asia, and Oceania (except those included in developed countries). Eastern Europe + FSU include the 14 countries of central and eastern Europe and all 15 countries of the Former Soviet Union (FSU) (9).

growth will occur (7). Feed grains are projected to account for 35% of the increase in global cereal production to 2020 because the majority of increase in meat production will most likely come from grain-fed poultry and swine produced in confined feeding operations, which require about 3 kg grain to produce 1 kg of meat (8). Therefore, world cereal demand is projected to increase by about 1.3% annually to 2025 (Table 1).

To evaluate prospects for conserving natural resources and improving environmental quality while meeting increased food demand, we interpret recent trends and future trajectories in land use, crop yields, nitrogen fertilizer use, carbon sequestration, and greenhouse gas emissions to identify key issues and challenges. Our discussion will emphasize cropping systems that produce maize, rice, and wheat because these three cereals provide about 60% of all human calories, either directly as human food or indirectly as feed grains for livestock, and will likely remain the foundation of the human food supply because of their high yield potential

**TABLE 1** Projected changes in population, cereal demand, yields, area, and prices from 1995 to 2025. Values shown refer to the business-as-usual scenario of food and water demand and supply based on the International Model for Policy Analysis of Agricultural Commodities and Trade (IMPACT) model (2). Population projections are based on the medium scenario of the United Nations' 1998 projection (70)

| Indices | 1995 | 2025 | Annual rate of change (%) |
|---|---|---|---|
| Global population (billion) | 5.66 | 7.90 | 1.12 |
| Global demand for rice, wheat, and maize ($10^6$ Mg)[a] | 1657 | 2436 | 1.29 |
| Total rice, wheat, and maize area ($10^6$ ha) | 506 | 556 | 0.31 |
| Mean rice, wheat, maize yield (Mg $ha^{-1}$)[a] | 3.27 | 4.38 | 0.98 |
| World rice price (U.S. dollars $Mg^{-1}$, milled rice) | 285 | 221 | −0.84 |
| World wheat price (U.S. dollars $Mg^{-1}$) | 133 | 119 | −0.37 |
| World maize price (U.S. dollars $Mg^{-1}$) | 103 | 104 | 0.03 |

[a]Numbers for cereal demand and yields are higher than those published in Rosegrant et al. (2) because rice is included as rough rice (paddy).

and ease of storage and transport. We will not cover the availability of water for irrigation, which is another critical resource for cereal production, because several recent reviews provide a thorough examination of the issues related to freshwater supplies for irrigated agriculture (2, 10–12).

## ARABLE LAND RESOURCES

### Estimating Land Reserves

Assessment of land resources available for agricultural expansion is estimated by the difference between land area currently used for crop production and land area that has the potential to produce crops. Most recent assessments rely on the land and crop database of the United Nations Food and Agriculture Organization (FAO) (9). Based on this approach, arable land reserves are estimated to be at least equal in size to the present area of cultivated land. For example, total land area suitable for production of at least one food crop was estimated at 3325 million ha (Mha), while existing irrigated and rain-fed cropland was estimated at 1505 Mha in 1994–1996, with another 156 Mha in settlements, roads, and infrastructure (13). By difference, the land reserve available for crop production was estimated at 1664 Mha, which is larger than the amount of cropland in current production.

Given this seemingly large land reserve, econometric models developed to predict future food demand-supply scenarios are typically based on the assumption that availability of arable land is not a constraint to expansion of cropped area (14). Instead, cereal prices have the greatest influence on cropped area expansion. One such model is the International Model for Policy Analysis of Agricultural Commodities and Trade (IMPACT) model developed by the International Food Policy Research Institute, which projects a 10% increase (50 Mha) in harvested cereal area from 1995 and 2025 (Table 1), an increase that is about double the current U.S. maize area. Expansion of cereal production area in sub-Saharan Africa and South America is expected to account for most of this increase. Because this expansion is associated with constant or declining cereal prices, the increase in cropped area would be much greater if cereal prices were to rise.

In contrast to this reassuring scenario of surplus arable land, Young (15) argues that the difference method is grossly misleading because it overestimates the amount of uncultivated land that can be farmed in a sustainable fashion, underestimates the amount of land currently in crop production, and neglects the increasing demand for land used for nonagricultural purposes. For example, annual cereal cropping currently practiced on steeply sloping cropland in South and Central America, the Great Lakes region of Central Africa, and in southern China is not likely to be sustainable over the longer term because of severe erosion risk. Perennial crops and agroforestry systems are better suited to these environments. Likewise, sustained cereal production is questionable in the semiarid zones of sub-Saharan Africa where population pressure has forced increased cropping intensity on soils of low fertility.

## Case Studies: Sub-Saharan Africa and China

A critical issue for sub-Saharan Africa is whether food demand can be met by intensification of crop production on existing cropland without further expansion of agriculture into more marginal production areas where the risk of crop failure, soil degradation, and environmental damage is high. Unfortunately, FAO statistics provide little insight into this issue because only harvested area is reported. For example, the increase in harvested crop area accounted for nearly all of the increase in food production in sub-Saharan Africa between 1989–1999, when there was little increase in yield of the major food crops (Table 2). It is impossible, however, to determine the relative contributions of intensification from growing two or more crops per year on the same piece of land in subhumid and humid areas, versus a reduction in length of the fallow period in semiarid zones, or actual expansion of cropped area into previously uncultivated areas. Lack of such data makes it difficult to estimate available land reserves to support sustainable crop production in sub-Saharan Africa.

Cultivated land area also is underestimated in some highly productive regions. Recent estimates of cropland in China, confirmed by remote sensing, are 35%–40% greater than the cultivated area reported in official government land-use statistics (16). But even with this larger estimate of cropped area, the land difference method suggests an additional land reserve of 30 Mha suitable for grain production. Such estimates do not account for the areas currently in cereal production systems that are not sustainable and the increasing amount of land needed for purposes other

**TABLE 2** Annual percentage rates of change in area, yield, and production of the major food crops in sub-Saharan Africa, 1989–1999[a]

| Crop | Area | Yield | Production |
|---|---|---|---|
| Cassava | 2.6 | 0.7 | 3.3 |
| Maize | 0.8 | 0.2 | 1.0 |
| Yam | 7.2 | 0.4 | 7.6 |
| Cowpea | 7.6 | −1.1 | 6.5 |
| Plantain | 1.9 | 0.1 | 2.0 |

[a]Annual growth rates in area, yield, and production were estimated from three-year means in 1988–1990 and 1998–2000 (9).

than agriculture. More than 2 Mha of agricultural land in China were converted to other uses from 1985 to 1995 (14). If China continues to follow trends in developed countries, the demand for national parks, recreation areas, and scenery will increase rapidly as economic development proceeds. Moreover, the process of industrialization and urbanization will continue to encroach on existing highly productive agricultural land while expansion of cropping will occur on more marginal land. Similar development processes will reduce arable land reserves for agriculture in other densely populated regions of South and Southeast Asia, which are currently major centers of cereal production. Assuming a requirement for housing and infrastructure of 40 ha per 1000 people, FAO estimates a need for an additional 100 Mha of urban land in developing countries by 2030 (14).

## Preserving Natural Ecosystems

Native forests, savannas, and wetlands account for a large portion of the remaining land reserves worldwide. Forests currently occupy about 27% of the uncultivated land in South and Central America and Africa (13). Preserving a large portion of these forest ecosystems is crucial for protecting the biodiversity and environmental services they provide. In addition, much of the remaining uncultivated land has severe constraints to crop production from soils that have physical or chemical properties that would limit plant growth without ameliorative amendments. Recent estimates suggest that only 7% and 12% of the potentially arable land in Africa and Latin America, respectively, are free of soil constraints (1). Sustaining productivity on such land requires proper soil management technologies and improved use of nutrients and other amendments to maintain soil quality. It should also be noted that the current status of land degradation is not precisely known because the most comprehensive survey to date, the Global Assessment of Land Degradation (GLASOD), was conducted more than 12 years ago (14). The GLASOD assessment estimated the total area of degraded land to be 1964 Mha, with nearly half of this area degraded to at least a moderate degree. Most of this degradation was the result of inappropriate agricultural practices.

Recent trends indicate that total harvested cereal area has been decreasing since 1980 (Figure 1*b*). Although most of the decrease has occurred in developed countries, especially in Eastern Europe and the Former Soviet Union (FSU), harvested area has also decreased slightly in some developing countries since 1995. This trend raises the issue of whether the cost-benefit ratio of expanding cereal area in developing countries will be favorable if cereal prices continue a slow decline as predicted by the IMPACT model (Table 1). Given the uncertainty in the estimates of land reserves for sustainable crop production, the steady conversion of agricultural land to other uses, and the need to protect large tracts of natural ecosystems, it seems prudent to establish policies at national and regional levels that minimize expansion of agriculture into uncultivated areas by meeting increased food demand with greater yields on existing cropland (17). Increased yields, however, depend on maintaining an exploitable yield gap and the use of management practices that maintain soil quality and reduce the negative effects of crop cultivation on environmental quality.

## YIELD POTENTIAL AND EXPLOITABLE YIELD GAPS

### Definitions

Yield potential is defined as the yield of a crop cultivar when grown in environments to which it is adapted, with nutrients and water nonlimiting and pests and diseases effectively controlled (18). Hence, for a given crop variety or hybrid in a specific field environment, yield potential is determined by the amount of incident solar radiation, temperature, and plant density—the latter governs the rate at which the leaf canopy develops. The difference between yield potential and the actual yield achieved by farmers represents the exploitable yield gap (Figure 2). Yield potential can be reduced by insufficient water supply, either from inadequate rainfall in rainfed cropping systems or from suboptimal irrigation in irrigated systems. Hence, genotype, solar radiation, temperature, plant population, and degree of water deficit determine water-limited yield potential. In addition to yield reduction from limited water supply, actual farm yields are determined by the magnitude of yield loss from factors such as nutrient deficiencies or imbalanced nutrition, diseases, insect pests, and weed competition.

As average farm yields approach the yield potential threshold, it becomes more difficult for farmers to sustain yield increases because further gains require the elimination of small imperfections in the integrated management of soil, crops, water, nutrients, and pests. In general, such rigorous fine-tuning is not economically viable on a production scale such that yield stagnation typically occurs when average farm yields reach about 80% of the yield potential ceiling (20).

### Importance of Maintaining an Exploitable Yield Gap

Lack of an increase in rice yield potential is a mounting concern because yield stagnation is occurring in some of the world's most productive rice-producing

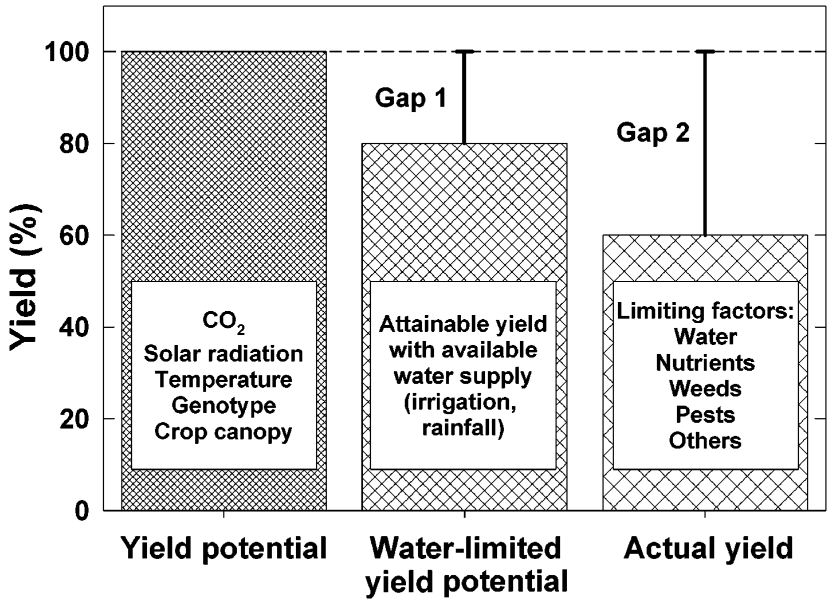



**Figure 2** Conceptual framework of yield potential, water-limited yield potential, and actual farm yields as constrained by a number of production factors. Modified from van Ittersum & Rabbinge (19).

regions as a result of a diminishing exploitable yield gap. Although aggregate rice yields in China appear to continue at a linear rate of increase established during the past 35 years (Figure 3*a*), yields are now approaching the 80% yield potential threshold in several major rice-producing provinces. For example, clear trends of yield stagnation are evident in three of China's major rice-producing provinces, which account for more than 35% of Chinese rice production (Figure 3*b*). Likewise, yields are increasing very slowly in Japan (Figure 3*a*) and Korea (data not shown), where average farm yields are currently about 80% of yield potential estimated by crop simulation models (21).

Yields are also stagnating in major rice-producing provinces of India (Punjab), the Philippines (Central Luzon), and Indonesia (Central Java), although these yield plateaus appear to be well below the 80% yield potential level (Figure 3*c*). Substantial reductions in yield growth at levels below the 80% yield potential threshold are typical of trends observed in a number of other regions and countries where modern rice production technologies have been practiced for several decades. Specific reasons for the yield stagnation in these regions have not yet been identified due to a lack of long-term monitoring data on biophysical and socioeconomic determinants of yield and productivity (22). Because yield stagnation in these areas is not associated with a diminishing exploitable yield gap, available evidence suggests productivity constraints from factors such as deterioration of soil and water

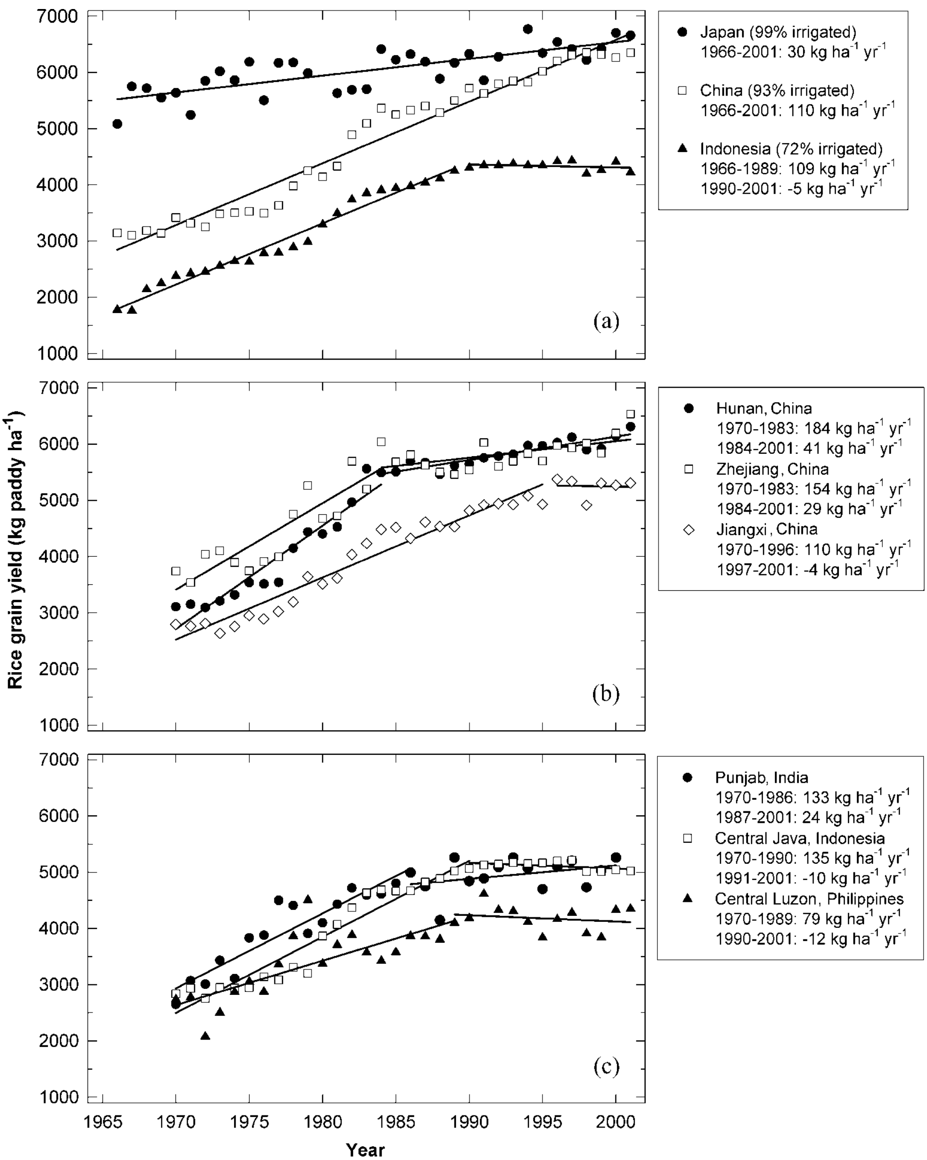



**Figure 3** Yield trends in major rice-producing countries and provinces where there is evidence of stagnation in the rate of gain in average rice yields. Country data obtained from (9). Province data were based on national agricultural statistics provided by D. Dawe, Social Sciences Division, International Rice Research Institute (IRRI). Note that yield data for China refer to official statistics. Actual yields are likely to be lower because of the apparent underestimation of crop harvest area in China (20).

quality, reduced access to irrigation water, and imbalanced nutrient use. It is also noteworthy that researchers have found it difficult to maintain yields at 80% of yield potential in all but a few of the long-term experiments on intensive irrigated rice systems in the developing countries of Asia (23).

## Estimating Trends in Rice Yield Potential

Maintaining an exploitable yield gap as average farm yields approach 80% of yield potential depends on achieving increases in yield potential through genetic improvement. Estimating trends in crop yield potential over time, however, is not a straightforward proposition. The most common method compares a time series of historical cultivars in a replicated field study. Cultivars chosen for such evaluations are typically the most widely used commercial varieties or hybrids of their time. The change in yield potential is estimated by plotting the yield of each cultivar against its year of release. A significant positive slope between yield and year of release is assumed to estimate the gain in yield potential—assuming the experiment is grown under nonlimiting conditions. This method places older cultivars at a disadvantage, however, because they were selected to withstand pathogens, insects, and soil and atmospheric conditions that existed during the period in which they were selected. But pathogen and insect pest populations evolve to overcome a cultivar's resistance to infection or infestation, soil properties change with intensive cropping, and atmospheric temperature and [$CO_2$] have risen steadily during the past 50 years. Whereas newly released cultivars are selected against contemporary conditions and are adapted to withstand them, older cultivars were not. Therefore, even with the best possible management practices to minimize the confounding effects of selection under different environmental conditions, it is not always possible to fully protect and optimize growth conditions for older cultivars.

Such a scenario is representative of modern rice-breeding efforts for intensive rice systems in tropical Asia. When historical inbred *indica* rice cultivars were grown in a replicated field study at two sites in 1996, there was a positive linear relationship between yield and year of release since 1966, with a slope of 75 kg $ha^{-1}$ $yr^{-1}$ (Figure 4). The oldest cultivar in this time series is IR8, which was released in 1966 and was the first widely grown modern inbred *indica* rice variety in tropical Asia. Although IR8 had the smallest yield when grown in 1996, it often attained yields of 9–10 Mg $ha^{-1}$ when grown in the first years after it was released, and this yield level is comparable to the yield potential of the most recently released cultivars. The yield potential for this environment estimated by simulation is also 9–10 Mg $ha^{-1}$ in years with typical weather patterns (21, 24). Hence, there has been no detectable increase in the yield potential of inbred rice varieties in 37 years since the release of IR8 (25, 26). Despite lack of progress towards greater yield potential, maintenance breeding efforts were highly successful in improving grain quality and maintaining yields in the face of substantial increases in disease and insect pressure—accomplishments of tremendous importance to sustaining rice production increases in Asia without expanding crop area.

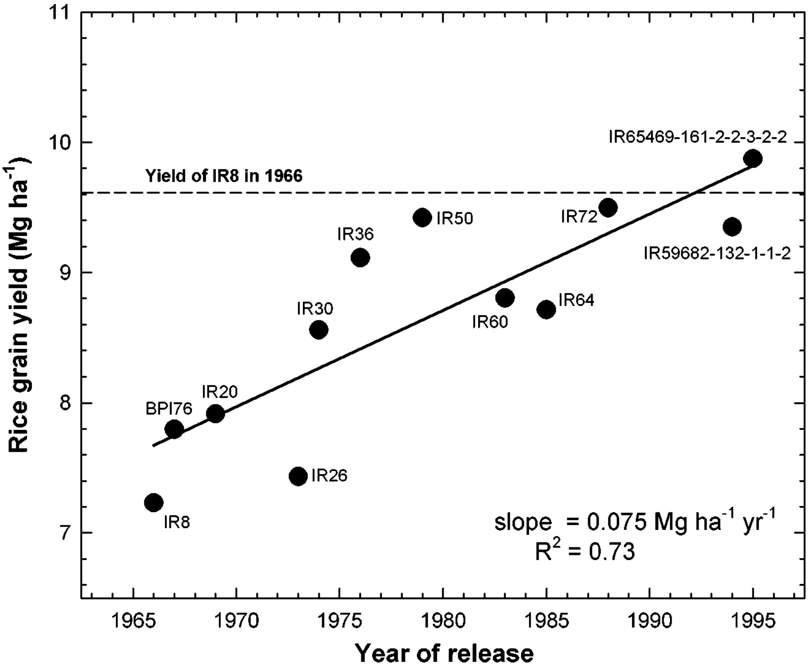



**Figure 4** Yield trend of cultivars and lines developed since 1966 at the International Rice Research Institute (IRRI) in the Philippines and by the Bureau of Plant Industry, Department of Agriculture, Philippines. The twelve cultivars and two lines were grown at the IRRI Research Farm and the Philippines Rice Research Institute Research Farm in the 1996 dry season with optimum crop management. Each data point is a mean of the two locations. The dashed line represents the maximum yield obtained with IR8 when grown at the IRRI Research Farm 30 years earlier, in the dry season of 1966 (28). The figure is modified from Peng et al. (27).

When older cultivars are at a disadvantage in a historical time series comparison, a positive slope in the relationship between cultivar yield and year of release provides an estimate of resistance to contemporary stresses rather than an estimate of yield potential as shown conceptually in Figure 5. Under this scenario, new cultivars of a given crop are released at regular intervals, and the yield of each cultivar declines as they become less adapted to evolving conditions in the agro-ecosystem. While maintenance breeding continuously identifies new cultivars with yield potential equivalent to older cultivars, there is no increase in yield potential per se.

Although there has been little, if any, improvement in yield potential of inbred *indica* rice varieties, there is convincing evidence of gains in yield potential from hybrid rice. Direct field comparisons of recently released *indica* rice hybrids with recently released inbred *indica* varieties have clearly documented that rice hybrids

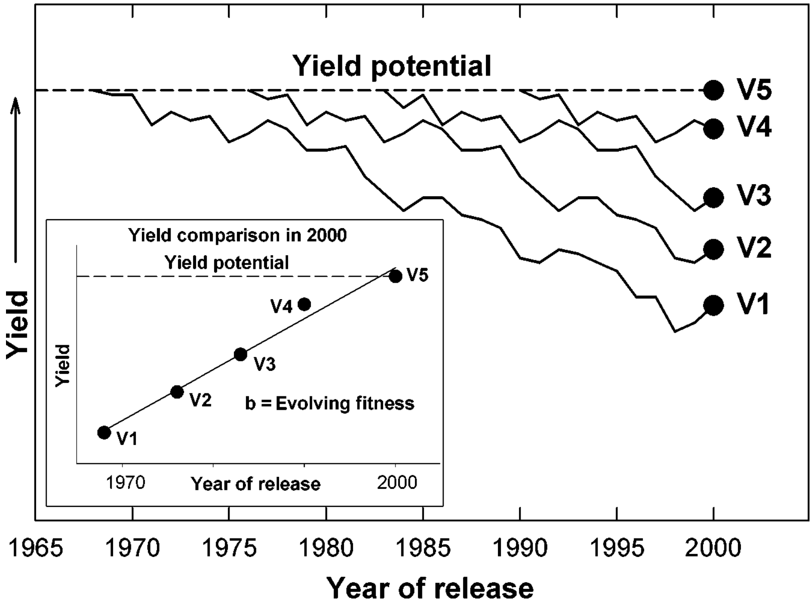



**Figure 5** Conceptual framework for breeding to maintain yields against evolving sensitivity to pathogens, insect pests, and abiotic environmental conditions without an increase in yield potential.

have a 9% advantage in yield potential when grown in tropical lowland environments (25). Hybrid rice presently accounts for about 50% of the rice area in China. Adoption of hybrids is beginning to occur in Vietnam, India, the Philippines, and Bangladesh, although they currently account for only 1.2% of global rice area outside China. Recent trends indicate that hybrid rice area in China has remained stagnant, and there are major impediments to commercialization in other countries. These impediments include the low yield of hybrid seed production, high seed cost, and poor grain quality of hybrid rice varieties. Moreover, the 9% yield potential advantage of hybrid rice represents a onetime gain from hybrid vigor rather than a sustained increase in yield potential over time.

In addition to hybrid rice, efforts are currently in progress to create new plant types with higher yield potential by crossing germplasm from tropical *japonica* germplasm with inbred *indica* varieties (26). These efforts follow upon a 10-year breeding program, beginning in 1990, which developed the tropical *japonica* germplasm into lines that were adapted to lowland tropical environments. With continued investment in this program, it may be possible to increase rice yield potential by 5%–10%. While an increase of this magnitude is far less than the 25%–50% increases that the International Rice Research Institute (IRRI) had initially hoped for, even a small boost should be considered a major accomplishment given the lack of increase in *indica* inbred rice yield potential since 1966.

## Estimating Trends in Yield Potential of Maize and Wheat

Estimating trends in maize yield potential is also difficult. Although maize breeders have been successful in developing hybrids with greater stress resistance, there is little evidence of an increase in yield potential (29). In part, the lack of evidence reflects the scarcity of research investment in maize yield potential in both the private and public sectors. It may also reflect the brute force empirical selection approach used by maize breeders, which relies on testing tremendous numbers of hybrid lines in thousands of on-farm strip trials with primary emphasis on yield and yield stability. Such on-farm trials rarely provide management practices that support yields that approach yield potential levels. The result has been substantial improvements in resistance to the wide range of abiotic and biotic stresses that occur under on-farm conditions and greater adaptation to intensified crop management practices adopted by farmers during the past 40 years.

Without explicit research efforts on maize yield potential, the highest reported maize yields come from nationally sanctioned yield contests that include hundreds of farmers who adhere to contest guidelines with regard to minimum field size, harvest area, and independent verification (30). Yield trends of contest winners for irrigated systems in the state of Nebraska indicate no increase in yield potential in the past 20 years, with a mean winning yield of 18.8 Mg $ha^{-1}$ (Figure 6). In contrast, contest-winning yields in rain-fed systems have increased markedly and are approaching the yield potential ceiling indicated by the irrigated contest-winning yields.[2] And while the current average irrigated maize yield is only 50% of yield potential, average yields are steadily increasing and will eventually approach the 80% yield potential threshold where stagnation occurs. In fact, a number of progressive maize farmers currently achieve yields that exceed 80% of yield potential in irrigated systems.

Investment in research to improve yield potential of wheat has been much greater than that for rice or maize. A number of field studies have compared yield trends of an historical time-series of wheat varieties (31–33). Although these studies suffer from the same confounding factors as comparable studies with rice, the wheat evaluations were more rigorous because yield gains were quantified with and without fungicide protection against diseases and at different levels of N fertilization. Results consistently document a linear increase in wheat cultivar yields versus year of release. The greater number of investigations and wider range of environments in which these tests were conducted give greater weight to evidence of genetic gain in wheat yield potential. Despite this apparent success, the rate of

[2]Although there are reports of considerably higher contest-winning yields at one site in Iowa, these yield levels are up to 50% greater than the yield potential simulated by existing maize models using actual data on climate, soil properties, planting date, and maturity of the hybrid used at the site (H. Yang, unpublished data). In contrast, the contest-winning yields in Nebraska fall comfortably within the range of simulated yield potential for these sites. Hence, we believe the contest-winning yields in Nebraska provide the most reliable estimate of maize yield potential in the north-central United States.

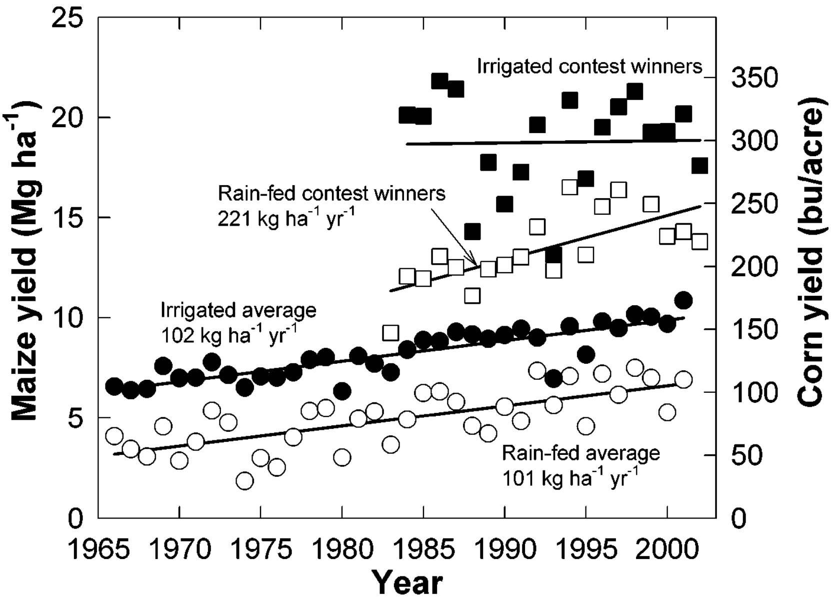



**Figure 6** Yield trends in yield contests sanctioned by the National Corn Growers Association for irrigated and rain-fed maize systems in Nebraska and average farm yields in Nebraska for irrigated and rain-fed maize production.

gain in these studies was estimated at 38 to 60 kg ha$^{-1}$ yr$^{-1}$, which is considerably less than 1% of the yield of the best-yielding cultivar. Because the rate of yield improvement was strongly linear in these studies, the proportional rate of gain will steadily decrease as yield potential increases. With annual wheat demand projected to rise by a compound annual rate of 1.1% to 2025, the exploitable gap between yield potential and average farm yields will also diminish in high-yielding wheat systems. Evidence of yield stagnation is apparent in the Yaqui Valley of Mexico, where the International Maize and Wheat Improvement Center (CIMMYT) conducts much of its wheat-breeding effort, and the linear yield trajectory in the Indian states of Punjab and Haryana will soon reach yield levels at which stagnation begins to occur in the Yaqui Valley (Figure 7). Together, the Punjab and Haryana states account for 34% of Indian wheat production. These trends emphasize the importance of continued efforts to increase wheat yield potential for sustaining yield gains at the farm level in major wheat-producing regions.

## Are Existing Yield Gaps Large Enough?

To answer this question requires estimation of mean crop yield potential in the most important cereal-producing areas worldwide and data on current yields in these areas. To estimate current crop yield potential requires a robust crop simulation model that has been validated against direct measurements of maximum attainable

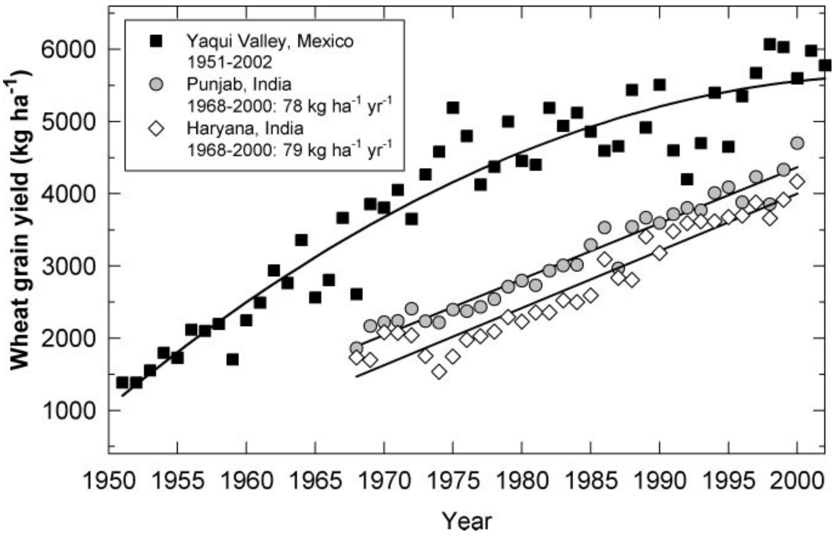



**Figure 7** Yield trends of wheat in the Yaqui Valley of Mexico and major wheat-producing provinces in India. Data for the Yaqui Valley were provided by K. Sayre, CIMMYT, and data for the Indian provinces were provided by D. Byerlee, World Bank.

crop yields from a number of environments and a long-term climate database for each of the major cereal-producing domains. Such geospatial modeling of yield potential and possible effects of different scenarios for global climate change have been evaluated for rice in Asia (21, 34). Based on this analysis, it is clear that yield potential must be increased to meet future demand projections without a large increase in rice production area because current irrigated rice yields have already approached or will soon approach 80% of current yield potential. As a result, yield stagnation is already occurring in many of the world's most productive rice domains (Figure 3*a*,*b*).

There has been no comparable effort to estimate maize and wheat yield potential in the most productive areas for these crops. Lack of such an analysis makes it difficult to estimate whether closure of existing yield gaps will meet projected demand for these cereals without expanding crop-production area. While the contest-winning U.S. maize yield typically ranges from about 15.7 to 20.0 Mg ha$^{-1}$ depending on year (30),[2] the average yield potential is much smaller because the contest-winning yield represents the highest possible yield achieved under the most favorable combination of soil, climate, and crop management. Current average U.S. maize yields are 8.6 Mg ha$^{-1}$ (1999–2001), which is perhaps 55% to 65% of the mean U.S. yield potential. Current average maize yields in developing countries, including China, are much smaller (2.96 Mg ha$^{-1}$ in 1999–2001) because of greater constraints from poor soils, water deficits, nutrient deficiencies, and pests in many production areas. Despite these constraints, numerous studies have shown considerable potential to increase yields with improved crop and soil management that includes nutrient input, weed control, and integrated pest management. Achieving adoption of such improved practices, however, will require substantial investment in applied research, extension, and market infrastructure—investments that are lacking in many developing countries.

Although it is clear that meeting projected rice demand will require both closure of the current exploitable yield gap and an increase in rice yield potential, the prognosis for maize and wheat is less certain. Our best guess is that closing the current yield gaps for maize and wheat is sufficient to satisfy demand for the next 20 to 25 years, but it will not be sufficient to meet the needs of a human population expected to reach 9 billion within 40 to 50 years. Hence, increasing yield potential of these cereals will also be a pivotal component of global food security.

## NITROGEN EFFICIENCY AND TRENDS IN NITROGEN FERTILIZER USE

Adequate N supply is required for achieving high cereal yields (35), but negative effects from improper N fertilizer use threaten environmental quality and human health at both local and global scales as a result of water pollution from nitrate leaching or runoff, air pollution, and greenhouse gas emissions. Estimates for the United Kingdom (36) and Germany (37) suggest that the environmental costs of N fertilizer use are equal to one third the total value of all farm goods produced. Because the relationship between crop yield and N uptake is tightly conserved (38), achieving further increases in grain production will require greater N uptake by these crops. Hence, the key challenge going forward is to meet the greater N requirements of higher-yielding crops while concurrently increasing N use efficiency and reducing the reactive N load attributable to agriculture. To address this challenge requires detailed understanding of crop response to N and reliable projections of cereal production increases at local, regional, and global scales.

### Inorganic Versus Organic Nitrogen Sources

Concern about the reactive N load from agriculture has led to calls for greater utilization of organic N sources and regulations reducing N fertilizer use. Organic production systems rely entirely on organic N sources. Even though only 1% of the world's cropland (about 16 Mha) is currently under certified organic production, demand for organic food is expected to grow, especially in developed countries, and organic agriculture may become a more widespread alternative to traditional agriculture in the next 30 years (14).

Although it is generally believed that organic agriculture offers environmental benefits associated with a reduction in pesticide use (14), the benefits from reliance on organic N sources have not been established, and the scientific basis for such a perception has not been documented. Controlling the fate of N from organic sources is just as difficult as managing the fate of mineral N fertilizer (39).

For example, nitrate leaching or runoff occurs whenever nitrate accumulation in soil coincides with a period of high rainfall or irrigation. Incorporating grass and legume cover crops, long fallow periods, mineral or organic N applications at the wrong time of the year, or small plant N demand (poor crop growth) can result in large nitrate leaching losses (40, 41). Case studies have shown comparable or increased potential losses (42), or decreased potential losses (43, 44) due to nitrate leaching and runoff from organic N sources as compared to fertilizer N. Many of these comparisons are flawed, however, because they compare different cropping systems, different amounts of applied N, and different yield levels. Under similar cropping systems with equivalent N input levels, nitrate losses from organic systems in the United Kingdom were similar to or slightly smaller than those from conventional farms following best management practices (45). Overall, the available literature provides no clear evidence that nitrate losses are reduced by the introduction of organic farming practices if the goal is to maintain the same crop yield levels as conventional farming systems (46).

Similar principles apply to other potentially harmful N loss mechanisms, such as gaseous N losses. In a study on a cultivated organic soil in southern Germany, total annual $N_2O$-N losses were 4, 16, 20, and 56 kg $ha^{-1}$, respectively, for a fertilized meadow, a fertilized arable field, an unfertilized meadow, and an unfertilized arable field (47). Although conversion from conventional to organic farming can sometimes reduce $N_2O$ emissions on an area basis, both systems emit similar amounts of $N_2O$ per unit of harvested yield (48, 49). In irrigated rice systems where rice is typically grown in flooded soil, methane emissions increase with the addition of manure and straw compared to systems that only receive mineral fertilizer N (50, 51). Reduction of N losses is therefore not a question of organic or conventional farming, but rather of using appropriate N management practices tailored to the needs of the particular cropping system.

Yield reductions are often associated with agricultural systems that follow organic practices (52, 53), and these systems appear to require both premium prices and government subsidies to remain economically viable. They also require copious amounts of organic N sources or increased land requirements to accommodate rotations with leguminous green manures to provide an adequate N supply. While this is feasible in industrialized countries, organic or low-input agriculture cannot secure the future food supply in the developing world, where maintaining low food prices contributes most to reducing poverty and increasing economic wealth (54, 55). Whereas organic N sources are critical components of the agricultural N cycle and should be utilized when they are available and cost-effective, cereal production at a global scale will largely depend on mineral N fertilizer to meet current and future food demand.

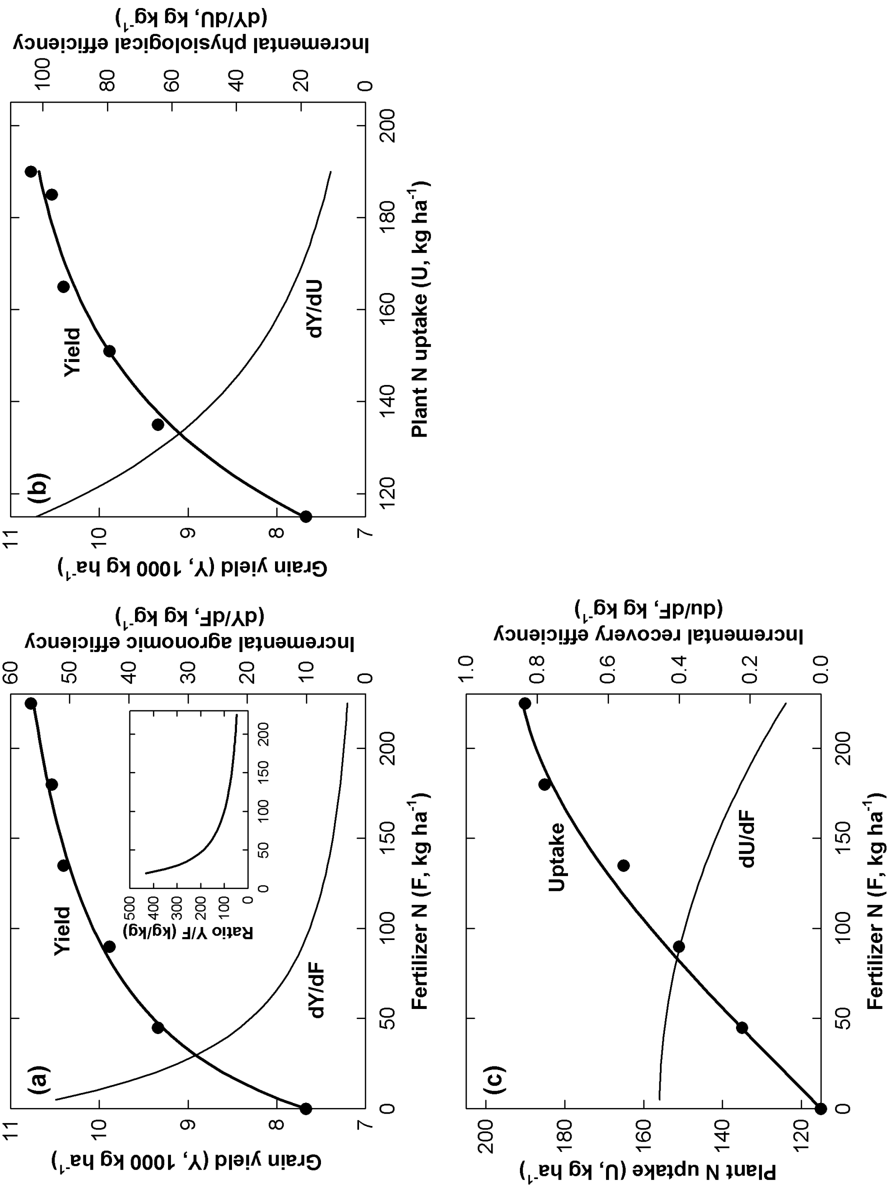
## Nitrogen Efficiency at the Field Level

The relationship between crop yield and N supply follows a diminishing return function that makes it difficult to achieve high yields and high N efficiency without increasing nitrate leakage or $N_2O$ emissions (38). At the scale of an individual field or experimental plot, grain yield (*Y*) and N uptake (*U*) increase with increasing N rate (*F*) and gradually approach a ceiling, which is determined by the site yield potential (Figure 8*a*,*c*). At low levels of N supply, rates of increase in yield and N uptake are large because N is the primary factor limiting crop growth and final yield. As the N supply increases, incremental yield gains are smaller because yield determinants other than N become more limiting. The broadest measure of N use efficiency is the ratio of yield to the amount of applied N (NUE $= Y/F$, also called the partial factor productivity of applied N), which declines from large values at small N rates to much smaller values at high N application rates.

Crop yield response functions to N vary widely among different environments, and they can be shifted due to technological, environmental, or economic factors (56). For example, the introduction of improved varieties with better adaptation to stress or innovations in N fertilizer management that improve the timing of N applications will shift the fertilizer response function up, which results in greater yield at the same level of N input (increase in NUE). Factors such as insufficient water supply, a decline in the indigenous N-supplying capacity of soil, a decrease in the uptake capacity of the root system due to soil toxicities or pathogens, yield limitations from deficiencies of nutrients other than N, and yield losses from insects, disease, and weeds can shift the response function down (decrease in NUE).

Nitrogen use efficiency is an aggregate efficiency index that incorporates the contributions from indigenous soil N, fertilizer uptake efficiency, and the efficiency with which N acquired by the plant is converted into grain yield. Evaluation of NUE requires separation of this aggregate efficiency index into component indices to understand the factors governing N uptake and fertilizer efficiency, to compare NUE in different environments, and to assess the effects of different N management options. To evaluate these components, agronomists typically estimate agronomic (AE), recovery (RE), and physiological (PE) efficiencies from applied N based on differences in yield and N uptake between fertilized plots and an unfertilized control (57, 58). Alternatively, the continuous response functions between yield, plant N uptake, and fertilizer N input illustrate the curvilinear nature of crop response to N application (Figure 8*a*,*b*,*c*). The incremental yield increase that results from N application can be defined as the incremental agronomic efficiency from applied N ($AE_i = dY/dF$ in Figure 8*a*). The $AE_i$ is the product of the efficiency of N recovery from applied N sources (incremental recovery efficiency, $RE_i = dU/dF$ in Figure 8*c*) and the efficiency with which the plant uses each unit of N acquired from applied N to produce grain (incremental physiological efficiency,

**Figure 8** Relationships among grain yield, plant N accumulation, and the amount of applied N in irrigated maize and their effects on different components of N use efficiency. Measured values (*symbols*) and fitted curves are based on a field experiment conducted in eastern Nebraska, which represents a favorable environment with fertile soils, use of a well adapted hybrid, and good pest control.


(a)
Yield
dY/dF
Ratio Y/F (kg/kg)
Grain yield (Y, 1000 kg ha-1)
Fertilizer N (F, kg ha-1)
Incremental agronomic efficiency (dY/dF, kg kg-1)
(b)
Yield
dY/dU
Grain yield (Y, 1000 kg ha-1)
Plant N uptake (U, kg ha-1)
Incremental physiological efficiency (dY/dU, kg kg-1)
(c)
Uptake
dU/dF
Plant N uptake (U, kg ha-1)
Fertilizer N (F, kg ha-1)
Incremental recovery efficiency (du/dF, kg kg-1)

$PE_i = dY/dU$ in Figure 8*b*). The $RE_i$ largely depends on the degree of congruence between plant N demand and the available supply of N from applied fertilizer or organic N sources. Consequently, optimizing the timing, quantity, and availability of applied N is the key to achieving high $RE_i$.

In addition to N uptake by the crop and N losses, a portion of the N from applied fertilizer and organic sources is retained in soil as residual inorganic N (either ammonium or nitrate) or incorporated into various organic N pools; these include microbial biomass and soil organic matter. Such retention should be considered a positive contribution to N input efficiency only when there is a net increase in total soil N content. Because more than 95% of total soil N is typically found in organic N pools, an increase in soil organic matter (i.e., carbon sequestration) is required to achieve increases in total soil N. Sustained increases in organic matter in cropping systems practiced on aerated soils (e.g., maize- and wheat-based systems without irrigated rice) result in greater indigenous N supply from decomposition of the organic N pools, which can reduce N fertilizer requirements to maintain yields and thereby increase NUE (59, 60). In contrast, greater soil organic matter in continuous irrigated rice systems does not necessarily result in an increase in N mineralization because there is little relationship between soil organic matter content and indigenous soil N supply in anaerobic soils (61, 62). For cropping systems in which soil organic matter is declining over time, there is an additional loss of N above that from applied N fertilizer and organic N sources. This additional loss of N reduces NUE, and greater amounts of applied N are required to maintain yields.

## Global Trends in Cereal Production and Nitrogen Fertilizer Use

Aggregate data on global crop production and fertilizer N consumption have often been used to estimate agriculture's contribution to the reactive N load in the global N cycle (3, 17, 63). At a global scale, cereal yields (Figure 1*c*, slope $= 45$ kg $ha^{-1}$ $yr^{-1}$), cereal production (Figure 1*a*, slope $= 31 \times 10^6$ Mg $yr^{-1}$) and fertilizer N consumption (Figure 9*a*, slope $= 2$ Mt $yr^{-1}$) have increased in a near-linear fashion during the past 40 years and are highly correlated with one another. Recent estimates indicate that the three major cereals receive 56% of global N fertilizer use while other cereals account for an additional 8% (64).

In developing countries, cereal yields and production from 1960 to 2001 follow a linear trend (Figure 1*a*,*c*). At the beginning of this time series, N fertilizer use was very small and increased exponentially during the course of the Green Revolution, resulting in a steep, nonlinear decline in the ratio of yield:N fertilizer use over time (Figure 9*b*). The rapid increase in N fertilizer use followed the rapid adoption of modern high-yielding varieties that could respond to the increased N supply and cropping intensity (66). The decrease in NUE occurs as farmers move yields higher along a fixed response function unless offsetting factors, such as improved management or yield limitations, shift the response function up or down (56).

In developed countries excluding those in Eastern Europe and the FSU, cereal yields continue to increase linearly (Figure 1*c*) while harvested area has declined

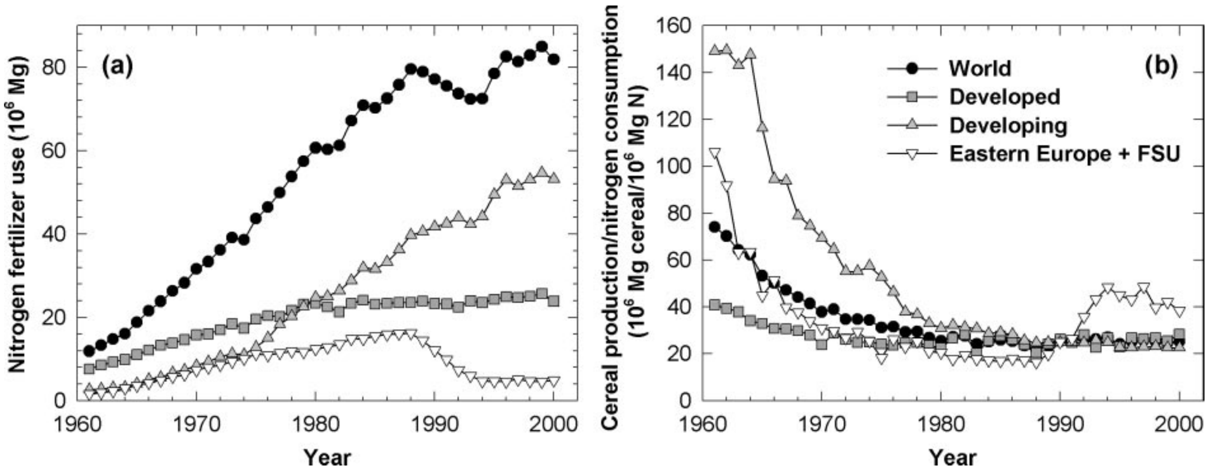



**Figure 9** Trends in global consumption of N fertilizer and the ratio of cereal production to N fertilizer consumption. *Developed countries* include those in Western Europe, North America, Australia, New Zealand, South Africa, Israel, and Japan. *Developing countries* include those in Latin America, Africa, Near East, Asia, and Oceania (except those included in developed countries). *Eastern Europe + the former Soviet Union (FSU)* include the 14 countries of central and eastern Europe and all 15 countries of the FSU. Sources: Production data obtained from FAOSTAT (9). Fertilizer consumption data obtained from the IFADATA database (65).

since the 1980s (Figure 1*b*), and total production (Figure 1*a*) and N fertilizer use (Figure 9*a*) have remained relatively constant. In Eastern Europe and countries of the FSU, N consumption dropped in the late 1980s as a result of political and economic turmoil. In these countries, the ratio of cereal yield:N fertilizer use doubled from 1988 to 2000 without improvements in yield potential or major changes in N management, and the ratio is now greater than in developing countries (Figure 9*a*,*b*).

The fact that trajectories of cereal production and N fertilizer use in developing and developed countries deviate from linearity is hidden in trends estimated from aggregate global data. Hence, the regression of global cereal production on global N use (Figure 9*b*) represents a crude index of global N use efficiency because this relationship is affected by changes in land area and yield, by stage of economic development, and by shifts in the yield response to N caused by adoption of improved germplasm and crop management technologies. Projections of future N fertilizer needs based on aggregated data can therefore be misleading unless these dissimilarities are considered.

## Disaggregating Trends in Cereal Yields and Nitrogen Fertilizer Use

The ratio of global cereal production to global fertilizer N consumption has been used to illustrate trends in NUE over time and shows a curvilinear decline in the past 40 years (Figure 9*b*). This decrease has raised concerns that future increases in N fertilizer use are unlikely to be as effective in raising yields as in the past (17). Aggregate global data, however, do not provide a sound basis for estimating future trends because these trends differ widely between developing and developed countries, as discussed above, between different countries, and among the different cereal crops.

The relationship between the mean national yield of maize, rice, and wheat and the mean rate of N fertilizer applied to each of these cereal crops on a country-by-country basis is linear (Figure 10). The slope (AE) of the combined regression suggests that cereal yields will increase by 37 kg $ha^{-1}$ for each kg of additional N fertilizer. The slopes and intercepts (Y at zero N applied), however, differ significantly among the three crops.

While the regressions in Figure 10 can identify major differences in N efficiency among crops or countries, they are of limited value for projecting future N fertilizer requirements because the combined regression includes countries with substantial differences in soil fertility and in the technological sophistication of crop management. Relationships between yield and N use within a country differ significantly from this global regression. For example, regression of average maize yield and N fertilizer rate for each of the major U.S. maize-producing states explains 26% of the variation in U.S. maize yield and has a slope (AE) of only 13 kg $kg^{-1}$ (Figure 11), which is nearly 70% less than the global AE for maize (Figure 10). In contrast, the U.S. regression has a large intercept of 6.1 Mg $ha^{-1}$, which is more than sevenfold greater than the global intercept because maize is generally grown on fertile soils

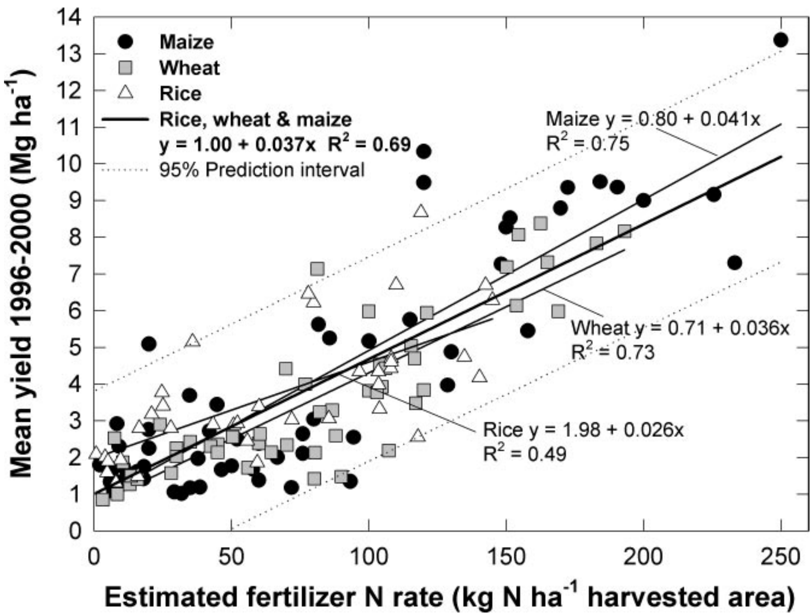



**Figure 10** Relationships between yield of maize, rice, and wheat and average N rates applied in each country. Each data point represents one country (rice, 37 countries; wheat, 53 countries; maize, 56 countries). Sources: yield data were obtained from FAOSTAT (9); fertilizer N rates represent country-specific estimates for each crop based on surveys and industry sources, as summarized in the 5th edition (2002) of the IFA/IFDC/IPI/PPI/FAO database on fertilizer use by crops (64). Values for each country refer to the average amount of N applied to the entire harvested area for each cereal crop.

in the U.S. Corn Belt. Increasing the N rate in the United States has relatively little effect on average yields because yield levels are already high and are approaching the nonlinear range of the N response curve (Figure 8*a*). Differences in the ratio of yield:N fertilizer rate among states are associated with substantial differences in soil quality and crop management. For example, the small yield:fertilizer N ratio in North Carolina is associated with the prevalence of highly weathered soils of relatively poor quality and small indigenous N supply in that state. In contrast, the larger yield:fertilizer N ratios in Wisconsin and Minnesota are associated with the higher quality of loess soils in those states, which have a greater indigenous N supply that shifts the N response curve upwards.

Relationships between yield and fertilizer N rate become even more scattered if farm-scale data are evaluated as seen in relationships among yield, plant N uptake, and fertilizer N rate from 179 fields under intensive rice cropping in Asia (Figure 12). These data are representative of much of the irrigated rice area in Asia. Average farm yields varied widely, but the mean yield of 5.1 Mg ha$^{-1}$ was

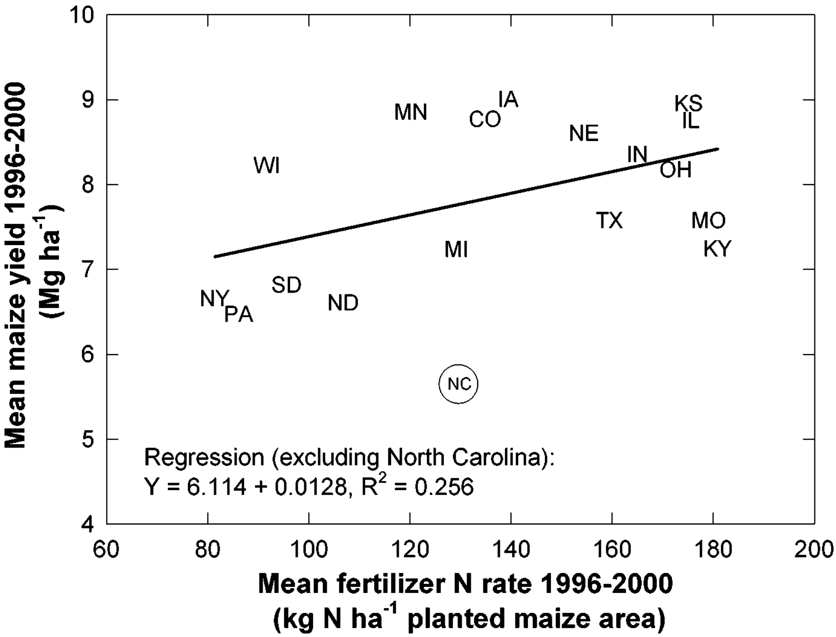
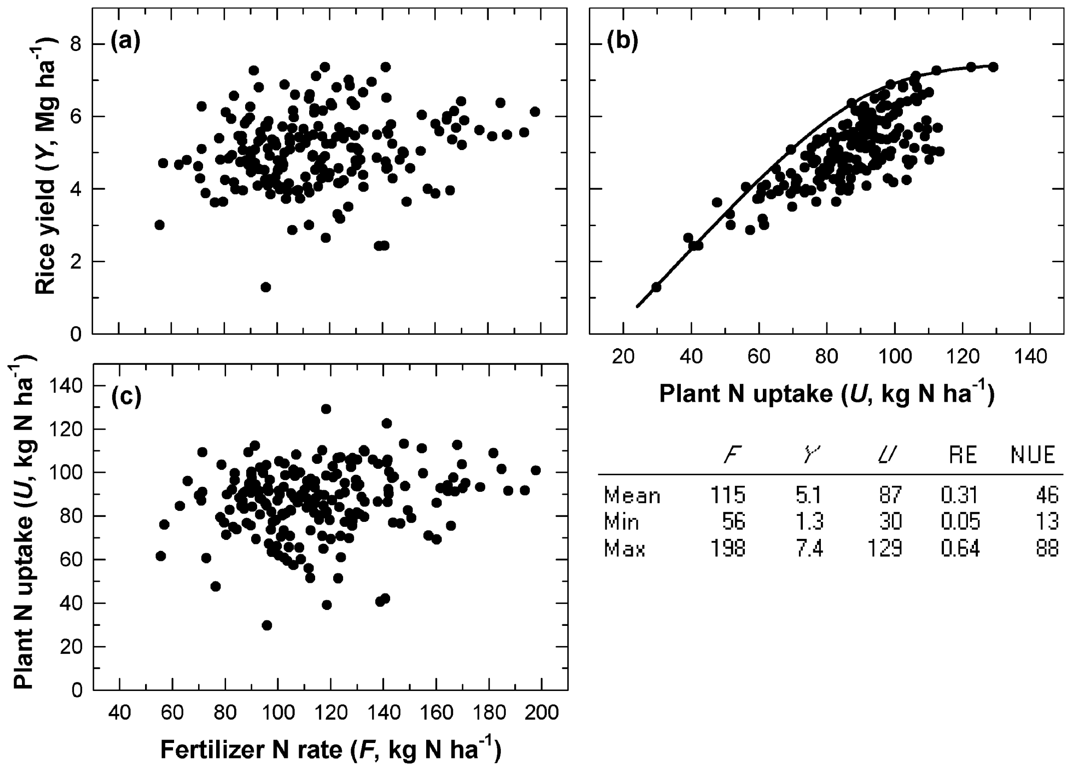



**Figure 11** Relationship between maize yield and N fertilizer rate in 18 U.S. states. Values shown are mean maize yields and the average fertilizer N use in each state during the period 1996 to 2000 (67).

close to the global average yield for irrigated rice of about 5.2 Mg ha$^{-1}$ (69). Average N rates applied by these farmers varied from 56 to 198 kg N ha$^{-1}$, but across farms there was no relationship between rice yield and N fertilizer rate, or between plant N uptake and N rate (Figure 12*a,c*). Yield was closely correlated with plant N uptake and formed an upper boundary line representing the most efficient N use for a given N rate at sites where N was the dominant yield-limiting factor (Figure 12*b*). This boundary line becomes nonlinear as yields approach high levels; which confirms the curvilinear nature of the relationship between yield and N uptake at the field level (Figure 8*b*). Numerous farms fell below this upper boundary, indicating that factors in addition to N limited yield, which explains the lack of a relationship between yield and N rate (Figure 8*a*). Among the factors that limit on-farm yields in addition to rate of N application are climate, the supply of other essential nutrients, disease, insect pest, weed pressure, stand establishment, and N management technology (e.g., timing, forms, and placement). As a result, the wide variation in NUE (46 to 88 kg grain kg$^{-1}$ per kg of applied N) was largely determined by the large variation in RE (0.05 to 0.64 kg plant N per kg of N uptake from fertilizer).

As a result of the large differences in NUE among countries, regions, farms, fields within a farm, and crop species, policies that promote an increase or decrease




| | *F* | *Y* | *U* | RE | NUE |
|---|---|---|---|---|---|
| Mean | 115 | 5.1 | 87 | 0.31 | 46 |
| Min | 56 | 1.3 | 30 | 0.05 | 13 |
| Max | 198 | 7.4 | 129 | 0.64 | 88 |

**Figure 12** Relationships among rice yield, N fertilizer rate, and plant N accumulation in 179 farmers' fields located in major irrigated rice areas of China, India, Indonesia, the Philippines, Thailand, and Vietnam. Data points represent means of individual rice farms (15 to 26 farms per country) for four consecutive rice crops grown from 1997 to 1999 based on data reported by Dobermann et al. (68).

in N fertilizer use at a state or national level would have a widely varying impact on yields, farm profitability, and environmental quality. Instead, achieving greater NUE at state or national levels will require policies that favor increases in NUE at the field scale with emphasis on technologies that can achieve greater congruence between crop N demand and N supply from all sources, which include fertilizer, organic inputs, and indigenous soil N (38).

## Projection of Future Nitrogen Fertilizer Requirements

Estimates of future growth in global fertilizer consumption differ because of different forecasting methods and underpinning assumptions about food demand, land area, yields, and trends in NUE. Because rice, wheat, and maize account for 56% of the global fertilizer N consumption, we evaluated nine scenarios of global fertilizer N consumption by the three major cereals to 2025 (Table 3). The different scenarios illustrate the sensitivity of N fertilizer requirements to trends in cereal yields, harvested area, and NUE.

At one extreme, scenario Ca, the harvested area declines by 7.4% due to decreasing availability of agricultural land at a rate similar to land use trends since 1980 (Figure 1*b*) and NUE decreases by 15%. Global N fertilizer consumption for rice, wheat, and maize in 2025 would be 61% larger in 2025 than in 2000 (1.9% $yr^{-1}$) under this scenario. The annual rate of yield increase (1.6%) must be large enough to meet increased food demand (1.3%) and offset the decrease in harvested area (−0.3% $yr^{-1}$). The average N rate to achieve the required yield level in 2025 is 151 kg N $ha^{-1}$, which is 74% above the current mean N rate. This large increase in N rate would increase environmental risks from N losses due to nitrate leaching, runoff, and $N_2O$ emissions because it is more difficult to match crop demand with N supply at higher rates of applied N.

Scenario Ab assumes that harvested cereal area increases at a rate of 0.3% $yr^{-1}$ from both area expansion and increasing cropping intensity. While this scenario is not consistent with global land-use trends of the past 20 years (Figure 1*b*), it is similar to the rate of increase in cultivated area predicted by the IMPACT model (2, 14). It is also assumed that NUE remains unchanged, although this will require continuing progress in crop genetics (increase in stress resistance and small increases in yield potential) and improved crop management technologies (reduction of yield gaps). Consequently, average N rates would rise by only 28% (1.0% $yr^{-1}$) and global N consumption would increase at the same rate as cereal production (1.3% $yr^{-1}$). The other scenarios with constant NUE (Bb, Cb) give equivalent projections for N fertilizer consumption because grain yield increases are proportional to increases in total N use. The assumption of constant NUE was also used to predict future N fertilizer consumption by the FAO baseline scenario (71), by the econometric model of Bumb & Banaante (72), and by Frink et al. (73) who based their projection on a model that considered population growth, gross domestic product, and crop production potential. Each of these models predicts an annual increase in N fertilizer consumption of 1.0% to 1.2% depending on the amount of expansion in cropped area in the different models.

**TABLE 3** Projected changes in N fertilizer requirements of the major cereals (rice, wheat, and maize) from 2000 to 2025 as affected by changes in harvested land area and N use efficiency (NUE in kg grain per kg of applied N fertilizer). All values refer to the sum or averages of global rice, wheat, and maize production[a]

| | **Change in harvested area[b]** | | | |
|---|---|---|---|---|
| | **A. Increase** | **B. Constant** | **C. Decrease** | **All** |
| | **Average N amount applied** | | | **Total N consumption** |
| **Change in N efficiency[b]** | **kg N ha$^{-1}$** | | | **$10^6$ Mg** |
| a. Decrease | 130.2 | 139.9 | 151.1 | 71.6 |
| b. Constant | 110.7 | 118.9 | 128.4 | 60.9 |
| c. Increase | 96.3 | 103.4 | 111.7 | 53.0 |
| | Cumulative change (%) | | | |
| a. Decrease | 50.2 | 61.4 | 74.3 | 61.3 |
| b. Constant | 27.7 | 37.1 | 48.1 | 37.2 |
| c. Increase | 11.1 | 19.3 | 28.8 | 19.3 |
| | Annual rate of change (%) | | | |
| a. Decrease | 1.6 | 1.9 | 2.2 | 1.9 |
| b. Constant | 1.0 | 1.3 | 1.6 | 1.3 |
| c. Increase | 0.4 | 0.7 | 1.0 | 0.7 |

[a]All scenarios assume that global population increases to 7.9 billion people (70) and that cereal demand increases by 37% (1.3% per year) to 2436 Mt in 2025 (2). Baseline global data for the year 2000 were a harvested area of 512 Mha, mean cereal yield of 3.47 Mg ha$^{-1}$, total production of 1777 Mt (averages for 1996 to 2000), and total fertilizer N consumption of 44.4 Mt (64). The global mean N fertilizer rate applied to rice, wheat, and maize was about 87 kg N ha$^{-1}$ in 2000 with a mean NUE of about 40 kg grain per kg N applied.

[b]*Area scenarios*: (A) Increase: Harvested area increases to 550 Mha in 2025 (+7.4%, +0.3% per year), mainly in sub-Saharan Africa and Latin America. Grain yield must increase to 4.43 Mg ha$^{-1}$ in 2025 (+28%, +1.0%/yr). This scenario is similar to the business-as-usual scenario in Rosegrant et al. (2). (B) Constant: Harvested area remains unchanged at 512 Mha. Grain yield must increase to 4.76 Mg ha$^{-1}$ in 2025 (+37%, +1.3%/yr). (C) Decrease: Harvested area decreases to 474 Mha in 2025 (−7.4%, −0.3% per year). Grain yield must increase to 5.14 Mg ha$^{-1}$ in 2025 (+48%, +1.6%/yr).

[c]*Nitrogen efficiency scenarios*: (a) Decrease: Insufficient investment in research that emphasizes improving crop management and increasing crop yield potential in favorable production areas such that NUE decreases to 34 kg grain kg$^{-1}$ applied N in 2025 (−15%, −0.6% per year). (b) Constant: NUE remains unchanged at 40 kg grain kg$^{-1}$ applied N. (c) Increase: Adequate investment in research that emphasizes improving crop management and increasing crop yield potential in favorable production areas such that NUE increases to 46 kg grain kg$^{-1}$ applied N in 2025 (+15%, +0.6% per year).

Scenario Ac is the most optimistic with regard to total N fertilizer requirements and minimizing negative environmental risks from the applied N because it assumes an increase in both area (+7.4%, or 0.3% yr$^{-1}$) and NUE (+15%, or 0.6% yr$^{-1}$). An increase in NUE could result from greater investment in research on genetic improvement and on research and extension to develop and implement improved crop and N management practices. This scenario gives a yield increase of 28% by 2025 (1.1% yr$^{-1}$) with an average N rate of only 96 kg N ha$^{-1}$ (+11%, or 0.4% $yr^{-1}$) as compared to present levels. The additional environmental risk from greater N fertilizer use would be minimized because of smaller N rates and a total increase in N fertilizer use of only 19% (0.7% $yr^{-1}$). Although this scenario minimizes environmental risk from N fertilizer, it increases the negative effects on natural resource conservation from expansion of cultivated area, especially if such expansion occurs at the expense of natural ecosystems or onto marginal land that cannot sustain intensive cereal production. Hence, Bc is perhaps the best overall scenario because it would increase NUE with no net change in harvested crop area and thereby achieve the required 37% increase in cereal production with a 19% increase in both N fertilizer use and N rate.

All of the scenarios of total N fertilizer use in Table 3 are much smaller than those from other studies that did not account for the interactive effects of changes in land area, yields, and NUE (66, 67, 77). Because N fertilizer requirements are sensitive to these factors and given the most likely trends in harvested area and NUE (Table 3), we believe our projections of global N fertilizer use on rice, wheat, and maize in 2025 (53 to 72 Mt N) represent a plausible range. Further improvements in predicting global N fertilizer use will require specifying the locations and cropping systems that will provide the increase in cereal production and the primary determinants of NUE in those environments.

## Improving Nitrogen Use Efficiency

The key question is whether the increase in NUE proposed in scenarios Ac, Bc, or Cc is realistic? We assumed a modest increase of 15% over a 25-yr period (0.6% $yr^{-1}$), which results in an average of NUE of 46 kg grain $kg^{-1}$ N applied as compared to 40 kg grain $kg^{-1}$ N at present. Far larger increases in NUE have been achieved in recent years in various developed countries. In U.S. maize systems, NUE increased from 42 kg $kg^{-1}$ in 1980 to 57 kg $kg^{-1}$ in 2000 (38), which represents a 36% increase (1.6% $yr^{-1}$). Three factors contributed to this improvement: (a) increased yields and more vigorous crop growth associated with greater stress tolerance of modern hybrids (29), (b) improved management of production factors other than N (conservation tillage, seed quality, and higher plant densities), and (c) improved N fertilizer management (74). In Japan, NUE of rice remained virtually constant at about 57 kg $kg^{-1}$ from 1961 to 1985 but has increased to more than 75 kg $kg^{-1}$ (32%, 1.8% $yr^{-1}$) in recent years (75, 76). Key factors contributing to this increase were a shift to rice varieties with better grain quality, which also had lower yield potential and nitrogen concentrations, and the adoption of more knowledge-intensive N management technologies (75); this resulted in a 17% decrease in the average N rate without a reduction in yield.

Increasing NUE in the developing world presents a greater challenge. Nitrogen use efficiency is particularly low in intensive irrigated rice systems of subtropical and tropical Asia, and the available evidence suggests that NUE has remained virtually unchanged during the past 20 to 30 years, despite increases in yield over time (74). Carefully conducted research has demonstrated, however, that rice is capable of taking up fertilizer N very efficiently (77) provided the timing of N applications is congruent with the dynamics of soil N supply and crop N demand (78). These principles have recently become embedded in a new approach for nutrient management (79, 80). In field testing conducted in 179 rice farms throughout Asia, average grain yield increased by 0.5 Mg $ha^{-1}$ (11%) and N fertilizer rate decreased by 5 kg N $ha^{-1}$ with field-specific management compared to the baseline farmers' fertilizer practice (68). Mean RE of applied N increased from 30% with farmers' practices to 40% with field-specific management that takes into account the large field-to-field variation in the indigenous soil N-supplying capacity. Studies in China documented even larger gains (81). Improving the congruence between crop N demand and N supply also were found to substantially increase N fertilizer efficiency of irrigated wheat in Mexico (82, 83).

These results highlight the potential for field-specific management in small-scale farming systems in developing countries, provided the technologies chosen match the biophysical and socioeconomic characteristics of the agroecosystem. Such improvements will require significant long-term investments in research and extension education. Several years of on-farm experimentation are required to develop an optimal N management scheme for a particular location that is characterized by a set of common environmental, socioeconomic, and cropping characteristics. Seasonal variation is large and fine-tuning of N management must be accomplished in accordance with other management factors that influence NUE, such as balanced supplies of macro- and micronutrients, water management, optimal plant density, and pest control. In addition, substantial investments in research to raise rice yield potential also will be required because many intensive rice-producing regions are currently approaching the yield potential ceiling (Figure 3), and the yield response to applied N becomes strongly curvilinear in this region of the response function. At present, we suspect that current investment in such research and extension efforts is grossly inadequate.

## CARBON SEQUESTRATION, GREENHOUSE FORCING, AND SOIL QUALITY

### Carbon Sequestration and Greenhouse Gas Emissions

The degree to which crop production systems contribute to increases or decreases in greenhouse gas concentrations is another issue of concern given modern trends in atmospheric composition and putative changes in global climate (84, 85). Over the past 52 years, atmospheric $CO_2$ concentrations have risen by 18%, which may contribute to global warming (86). Soil represents one of the largest pools of carbon (C) in the terrestrial biosphere and contains about 1500 Pg C in organic forms, which is roughly three times the size of the biotic pool of C in terrestrial ecosystems (87). Hence, small changes in size of the soil organic C (SOC) pool have a dramatic effect on the atmospheric C balance. Although most of the increase in atmospheric $CO_2$ has been driven by accelerated fossil-fuel use, agricultural activity through deforestation and soil cultivation has contributed an estimated 55 Pg C loss from decomposition of SOC and release of $CO_2$ to the atmosphere during the past 150 years (86). Current estimated annual net SOC loss from land use change is in the range of 1 to 2 Pg C $yr^{-1}$, which occurs primarily in the tropics (88). Agriculture also contributes to greenhouse forcing through the emission of $N_2O$ associated with the application of N fertilizer and through the consumption of fossil-fuel energy in the manufacture, distribution, and use of agricultural inputs and machinery. For example, application of N fertilizer accounts for about 60% of the fossil-fuel energy consumed in the production of U.S. maize (89).

The net effect of a cropping system on greenhouse forcing potential can be estimated by accounting for all greenhouse gases emitted to the atmosphere or sequestered in soil or plant biomass. Such an analysis must consider the greenhouse gas emissions associated with all inputs and outputs used in the production system. Rates of $CO_2$ oxidation from SOC in a given cultivated field vary in relation to soil moisture and temperature regime, soil physical and chemical properties, the amount of carbon (C) input from crop residues, the chemical composition of organic C in these residues, and the degree of physical soil disruption (e.g., tillage). Consequently there is considerable potential to minimize the oxidation of existing SOC and to increase the inputs of crop residue-C through changes in crop and soil management. The potential for C sequestration in stable soil organic matter has been estimated in several studies (90–94). Estimates of C sequestration potential in U.S. crop agriculture range from 0.075 to 0.208 Pg C $yr^{-1}$, which is equivalent to 5 to 12% of $CO_2$ emissions from total U.S. fossil-fuel consumption (Figure 13).

Global estimates of the mitigation potential of C sequestration in agricultural soils are in the 0.4 to 0.6 Pg C $yr^{-1}$ range, or less than 10% of the current annual C emissions from fossil fuels (93). In general, comparison of C sequestration rates in forest and agroecosystems suggest that sequestration potential is greater in timber production systems. However, recent results indicate that rapid turnover of organic C in the litter layer and N limitations to primary productivity of forest ecosystems, which are typically located on relatively poor soils, may limit the potential size of forest C sinks (95).

The estimates of C sequestration in cropping systems are derived from direct measurements in long-term experiments and monitoring sites coupled with simulation and extrapolation of these point estimates to regional, national, and global scales. Decreased tillage intensity, reduced bare fallow, improved fertilizer management, crop rotation, and cover crops are factors identified as having the greatest potential to increase C sequestration (92, 96). From our view, however, these estimates of agricultural C-sequestration potential are constrained by two factors. First, they are based on cropping systems that give average yields with average crop management despite the fact that average yields and biomass accumulation of the major cereal crops have increased steadily due to genetic improvements in crop cultivars and improved management of soil and inputs (18, 97, 98). For

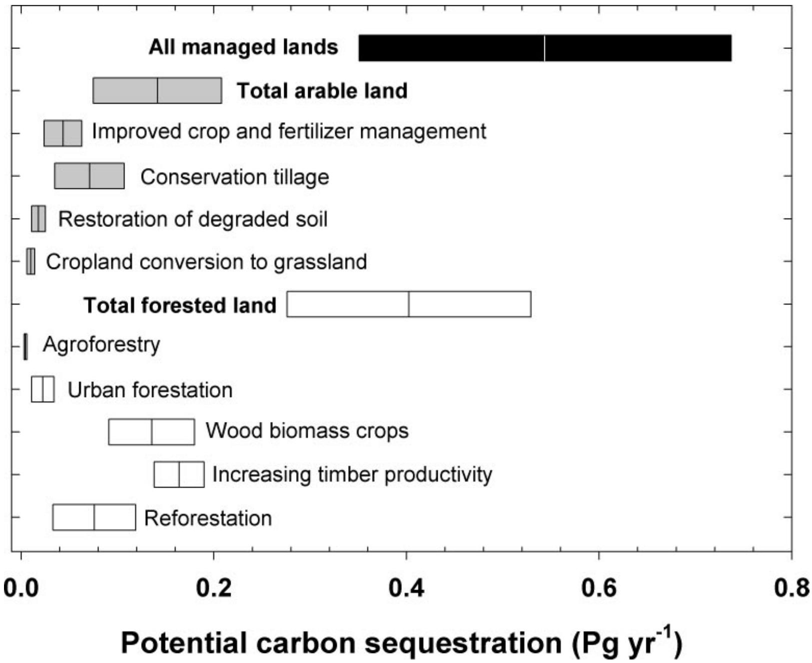



**Figure 13** Annual U.S. potential for C sequestration from the adoption of alternative land use options in managed forests and arable lands. Bars indicate current high and low estimates of potential C sequestration. Adapted from (85).

example, average U.S. maize yields have increased linearly for the past 35 years at a rate of about 109 kg $ha^{-1}$ per year. Moreover, many progressive U.S. farmers currently produce maize yields that are 55% to 75% greater than today's average farm yield. Similarly, comparisons of cropping systems are questionable when crop management is not clearly defined with regard to the yield potential of the each system. For example, West & Post (99) compared C sequestration as affected by changing from conventional tillage to no-till in 67 long-term experiments from across the United States. They reported a mean C-sequestration rate of 900 kg C $ha^{-1}$ $yr^{-1}$ for maize–soybean rotations ($n = 14$) but only 440 kg C $ha^{-1}$ $yr^{-1}$ for continuous maize ($n = 11$). Yet analysis of the same data set with respect to the overall rate of C sequestration in response to a change from continuous maize to maize-soybean gave a mean annual C-sequestration rate of $-190$ kg C $ha^{-1}$ $yr^{-1}$. Such discrepancies are likely to result from variation in the optimization of crop management and differences in sampling and measurement methods among these long-term experiments. Therefore, we believe that the most useful estimates of C-sequestration potential are derived from cropping systems managed to achieve yields that approach 80% of yield potential, which are attainable with progressive intensification strategies that increase both yields and input use efficiency. Such estimates would provide a more realistic prognosis for C-sequestration potential than estimates based on today's average yield with average management.

Second, the validity of C-sequestration assessments depends on the accuracy of estimated or simulated net primary productivity, the proportion of plant biomass that is returned to soil, and the rate of C transformations and turnover in soil. Several recent studies have attempted to assess current and future soil C-sequestration potential at a continental scale for Europe and North America based on ecosystem simulation models (89, 94, 100–105). Because these models are typically validated against data from long-term experiments in which net primary productivity and management follow current average practices, or even antiquated practices, their ability to simulate future scenarios outside the range of validation is questionable. For instance, three of the most widely used maize simulation models, CERES-Maize (106), Muchow-Sinclair-Bennett (107), INTERCOM (108), and the ecosystem C balance model CENTURY (109, 110), underestimated recycled aboveground crop residues by 13% to 47% when compared with field measurements in a high-yield long-term experiment at Lincoln, Nebraska (Figure 14). In addition to underestimation of crop residue yields, the CENTURY model overestimates root biomass. In one field study at Mead, Nebraska, measured maize root biomass was 1.9 Mg $ha^{-1}$ at anthesis, which is the point of maximum root biomass, but the CENTURY model predicted a root biomass more than threefold greater (D.T. Walters et al., unpublished data). Although current ecosystem C balance models are useful to explore future trends under different scenarios, more accurate and robust models are needed to provide reliable estimates of the actual magnitude of C sequestration in response to changes in crop management and climate—especially at crop yield levels that are substantially higher than current average yields of maize, rice, and wheat systems.

## Soil Quality, Nitrogen Requirements, and Greenhouse Gas Emissions

In addition to mitigation of greenhouse gas emissions, C sequestration benefits soil quality by increasing organic matter content (111, 112). Soil organic matter contributes to soil quality and ecosystem function through its influence on soil physical stability, soil microbial activity, nutrient storage and release, and environmental quality (113). The essential plant nutrients N, phosphorus (P), and sulfur (S) are components of the chemical building blocks that form soil humus. These nutrients are mineralized into plant-available forms by microbial activity that decomposes the humus. Humus is especially rich in N, which comprises 4%–6% of soil organic matter mass. Hence, C sequestration in soil humus requires input of both N and C that exceeds the output of these elements from an ecosystem.Thus, in many cropping systems, the application of N fertilizer increases soil C sequestration through augmented plant productivity and increased return of crop residues (114, 115).

When the net C and N balance results in C sequestration, the larger size of the SOC pool results in greater rates of SOC decomposition in aerobic soils. Because

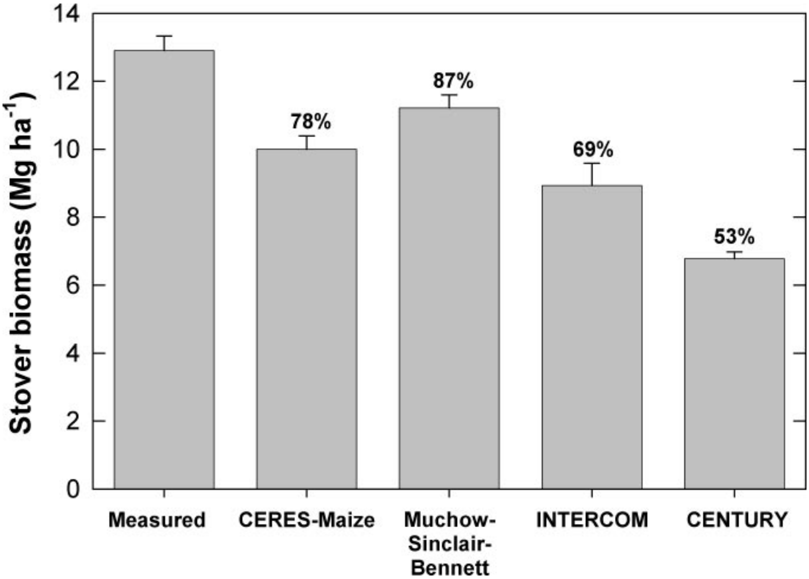



**Figure 14** Aboveground maize vegetative biomass (stover) at maturity as measured in a high-yield field experiment at Lincoln, Nebraska, and corresponding estimates of biomass yield by four widely used simulation models. Values shown are means and standard errors for three years (1999 to 2001) and three plant population treatments in each year ($n = 9$). Numerical values above simulation bars represent the percentage of the measured biomass yield. In this field study, the maize crop was managed to achieve the minimal possible stress from biotic and abiotic factors.

the C:N ratio of SOC is relatively stable, an increase in SOC decomposition will result in a greater indigenous supply of plant-available N and a reduction in N fertilizer requirements (scenario B, Figure 15). Additional improvement in NUE can occur when the benefits of C sequestration are coupled with increases in crop yields from adoption of cultural practices that reduce yield losses from abiotic and biotic stresses and improve N fertilizer efficiency (scenario C). Therefore, management practices and policies that encourage enhancement of soil quality through C sequestration will also lead to a reduction in N fertilizer requirements per unit of yield in cropping systems on upland (i.e., aerated) soils. The effect of enhanced soil quality from C sequestration also can improve crop yields from postive effects on other soil physical and chemical properties that influence root development, water-holding capacity, water infiltration rate, and the availability of P and S.

Nitrous oxide losses via nitrification and denitrification are estimated to average $1.25 \pm 1.0\%$ of applied N fertilizer (116, 117). Although this reference value is widely used to estimate $N_2O$ emissions from agriculture, the proportional loss

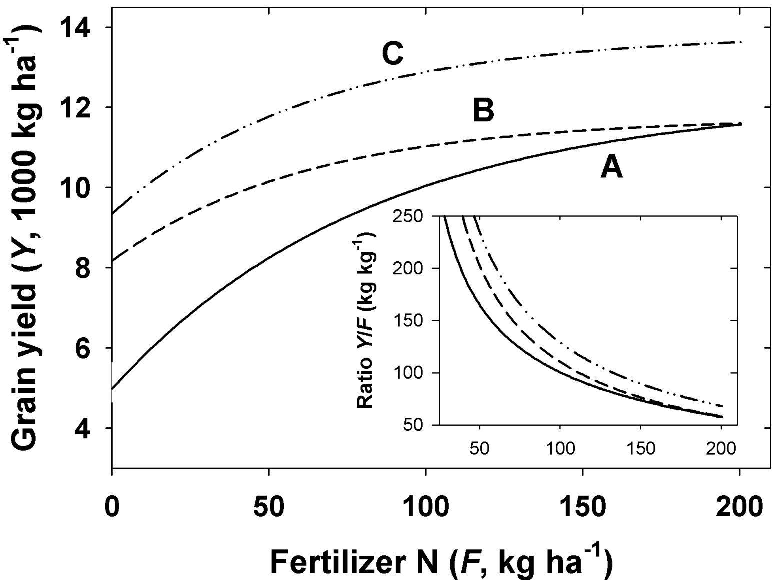



**Figure 15** Hypothetical relationship between maize yield (*Y*) and the N application rate (*F*) for average soil quality and average yield (*curve A*), average yield and increased soil organic matter content and associated indigenous N supply (*curve B*), and increased soil organic matter content and indigenous N supply with improved crop management to achieve greater N fertilizer efficiency at all rates of applied N (*curve C*). Scenarios B and C assume an increase of 50 kg N $ha^{-1}$ in indigenous N supply from the increase in soil organic matter. Insert shows the overall N use efficiency (*Y/F*) for each scenario.

may vary considerably as influenced by management practices, crop vigor, climate, and soil properties (118–120). Losses also differ among N fertilizer formulations with the greatest losses occurring from anhydrous ammonia (121). Despite this variation, denitrification losses are typically proportional to the amount of applied N fertilizer because the nitrification of $NH_4^+$ is associated with constituitive formation of small amounts of $N_2O$, and because nitrate is denitrified to $N_2O$ by anerobic reduction mediated by facultative microbes under wet soil conditions.

Recent field (122) and simulation studies (103) have demonstrated that trace gas fluxes and whole-system energy balance must be considered in quantifying the greenhouse forcing potential of different land management options. For example, maize-based cropping systems that dominate agricultural land use in the north-central United States are considered to have significant under-utilized C-sequestration potential (123), but they also contribute significantly to global greenhouse gas emissions. Hence, the positive effects of sequestering C can be offset by emissions of greenhouse gases such as nitrous oxide ($N_2O$) or inefficient use of fossil-fuel energy embodied in other crop and soil management operations (122). Previous research has illustrated effects of crop rotation, tillage, irrigation, crop residues, soil conditions (temperature, water status, pH, salt content, and available C), manure, and fertilizer use on emissions of $N_2O$ and other greenhouse gases from maize-based systems in the north-central United States (119, 121, 124–129). However, most of these estimates were obtained from small experimental field plots, which may not be representative of production-scale fields. Therefore, most currently used simulation models fail to account for large pulses of $N_2O$ emissions caused by spring thawing (130), rapid soil warming (131), tillage and irrigation events (132), and N application (120), which may greatly affect annual emission rates and the net global warming potential of an agroecosystem (133). And although C sequestration is often increased in systems that receive N fertilizer, the energy costs of N fertilizer and associated $CO_2$ emissions must also be included in the net greenhouse forcing budget (134).

In summary, a number of uncertainties exist about the design of optimal management practices to sustain increases in food production while optimizing N use efficiency and C sequestration and minimizing greenhouse gas emissions. Resolving these uncertainties will require carefully designed, interdisciplinary field studies to improve fundamental understanding of crop growth and C and N cycling in response to management and environment. This knowledge can then be used to develop robust ecosystem models that accurately simulate C and N balance across a wide range of environmental conditions. Greater emphasis on conducting such studies in production-scale fields with progressive management practices are also needed to obtain realistic estimates of future C-sequestration potential and effects on greenhouse forcing.

## CONCLUSIONS

A declining birth rate and projections for stable or decreasing human population within the next 40–50 years present an historical opportunity to protect natural resources for future generations. The degree to which agriculture contributes to resource conservation while meeting increased food demand is a critical component of this scenario. Increasing yields on existing cropland, limiting expansion of cultivated area, achieving a substantial increase in N fertilizer efficiency, and improving soil quality through C sequestration will be required to avoid severe natural resource degradation and to reduce emissions of greenhouse gases.

Although harvested cereal production area has remained relatively constant during the past 20 years, evidence of yield stagnation in several major cropping systems will make it increasingly difficult to sustain increases in food production without an expansion in cultivated area. Intensification of cereal production on existing cropland is required because loss of land to urbanization and industrialization largely occurs on prime agricultural land, and cropland replacement occurs at the expense of remnant forests and grasslands that typically have poorer soils and climate for intensive crop production. Lack of progress in raising yield potential is another threat to maintaining yield advances on existing agricultural land, and the scientific challenge of increasing crop yield potential appears to have been underestimated.

Intensification presents a challenge to reducing the negative effects of N fertilizer because crop yield response to applied N follows a diminishing return function at the field level. Organic farming is not a panacea because it is equally difficult to control the fate of N from organic N sources as it is from fertilizer, especially in systems that produce at equivalent yield levels. Technologies that improve the congruence between crop N demand and the N supply from soil and fertilizer have the greatest potential to improve N efficiency. Precise N management in time and space is required, which depends on accurate prediction of soil N supply and real-time crop N demand on a field-specific basis for small farms and a site-specific basis in large production fields. Significant strides have been made towards developing this capability, but continued investment in research and extension will be needed to assure practical management options and farmer adoption. Trends in NUE and cultivated area will ultimately determine global N fertilizer requirements and the risk of N losses to the environment.

The degree to which agriculture contributes to solving or aggravating atmospheric greenhouse gas composition depends on trends in soil C sequestration and NUE. Intensive cereal production systems appear to have considerable scope for sequestering C, which can reduce net greenhouse forcing potential when NUE is high and $N_2O$ emissions are low. Perhaps the greatest potential for short-term gain in C sequestration exists in the reversion of marginal lands currently under cultivation and unsuitable for sustainable production to native vegetation. Avoiding further expansion of agriculture into natural ecosystems is another key factor in limiting greenhouse gas emissions. Enacting policies to support the reversion of marginal lands and protection of natural ecosystems from agricultural expansion will place an additional burden on existing highly productive cultivated areas to meet future food demand. These same productive soils also have the greatest potential for C sequestration and increased NUE.

A number of influential crop scientists and economists see few technological or biophysical constraints to meeting global food requirements of an expanding human population (97, 135–137). These optimistic scenarios are based on two pivotal assumptions: (a) there is adequate arable land of sufficient quality to support increased grain production, and (b) an exploitable gap can be maintained between average farm yields and the genetic yield potential of the major cereal crops to allow sustained increases in crop yields. Our analysis suggests considerable uncertainty in both assumptions.

We conclude that an environmentally proactive agriculture will be required to meet food demand and protect natural resources and environmental quality. It will require policies and markets that direct intensification to existing prime agricultural land while avoiding expansion of cultivated area into natural ecosystems. It also will require substantial investments in research and extension to support scientific advances and timely development and adoption of innovative new technologies that help to close the exploitable yield gap, increase crop yield potential and N fertilizer efficiency, and improve soil quality.

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## LITERATURE CITED

1. Wood S, Sebastian K, Scherr SJ. 2000. *Pilot Analysis of Global Ecosystems: Agroecosystems*. Washington, DC: Int. Food Policy Res. Inst., World Resourc. Inst.
2. Rosegrant MW, Cai X, Cline SA. 2002. *World water and food to 2025: dealing with scarcity*. Washington, DC: Int. Food Policy Res. Inst. 338 pp.
3. Smil V. 1999. Nitrogen in crop production: An account of global flows. *Glob. Biochem. Cycles* 13:647–62
4. Dobson P, Bradshaw AD, Baker JM. 1997. Hopes for the future: restoration ecology and conservation biology. *Science* 277:515–22
5. Matson PA, Parton WJ, Power AG, Swift MJ. 1997. Agricultural intensification and ecosystem properties. *Science* 277:504–9
6. Vitousek PM, Mooney HA, Lubchenco J, Melillo JM. 1997. Human domination of Earth's ecosystems. *Science* 277:493–99
7. Rosegrant MW, Paisner MS, Meijer S, Witcover J. 2001. *Global Food Projections to 2020: Emerging Trends and Alternative Futures*. Washington, DC: Int. Food Policy Res. Inst.
8. Bradford E, Baldwin RL, Blackburn H, Cassman KG, Crosson PR, et al. 1999. *Animal agriculture and global food supply*. *Task Force Rep. 135* Counc. Agric. Sci. Technol. Ames, IA
9. Food Agric. Organ. UN. 2003. *FAOSTAT Database–Agricultural Production*. http://apps.fao.org
10. Pimentel D, Houser J, Preiss E, White O, Fang H, et al. 1997. Water resources: agriculture, the environment, and society. *BioScience* 47:97–109

11. Postel SL. 1998. Water for food production: Will there be enough in 2025? *BioScience* 48:629–37
12. Naylor RL. 1996. Energy and resource constraints on intensive agricultural production. *Annu. Rev. Energy Environ.* 21:99–123
13. Fischer G, Shah M, van Velthuizen H, Nachtergaele FO. 2000. *Global Agro-Ecological Assessment for Agriculture in the 21st Century.* Vienna: Int. Inst. Appl. Syst. Anal.
14. Food Agric. Organ. UN. 2002. *World Agriculture: Towards 2015/2030*. Rome: FAO UN. 97 pp.
15. Young A. 1999. Is there really spare land? A critique of estimates of available cultivable land in developing countries. *Environ. Dev. Sustain.* 1:3–18
16. Heilig GK. 1999. *China food: Can China feed itself?* CD-ROM Vers. 1.1 Laxenburg, Austria: Int. Inst. Appl. Syst. Anal.
17. Tilman D, Cassman KG, Matson PA, Naylor RL, Polasky S. 2002. Agricultural sustainability and intensive production practices. *Nature* 418:671–77
18. Evans LT. 1993. *Crop Evolution, Adaptation, and Yield.* Cambridge, UK: Cambridge Univ. Press. 500 pp.
19. van Ittersum MK, Rabbinge R. 1997. Concepts in production ecology for analysis and quantification of agricultural input-output combinations. *Field Crops Res.* 52:197–208
20. Cassman KG. 1999. Ecological intensification of cereal production systems: yield potential, soil quality, and precision agriculture. *Proc. Natl. Acad. Sci. USA* 96:5952–59
21. Matthews RB, Kropff MJ, Horie T, Bachelet D. 1997. Simulating the impact of climate change on rice production in Asia and evaluating options for adoption. *Agric. Syst.* 54:399–425
22. Dawe D, Dobermann A. 2001. Yield and productivity trends in intensive rice-based cropping systems of Asia. In *Yield Gap and Productivity Decline in Rice Production*, pp. 97–115. Rome: FAO UN
23. Dawe D, Dobermann A, Moya P, Abdulrachman S, Bijay S. et al. 2000. How widespread are yield declines in long-term rice experiments in Asia? *Field Crops Res.* 66:175–93
24. Kropff MJ, Cassman KG, Peng S, Mathews RB, Setter TL. 1994. Quantitative understanding of yield potential. In *Breaking the Yield Barrier*, ed. KG Cassman, pp. 21–38. Los Baños, Philipp.: Int. Rice Res. Inst.
25. Peng S, Cassman KG, Virmani SS, Sheehy JE, Khush GS. 1999. Yield potential trends of tropical rice since the release of IR8 and the challenge of increasing rice yield potential. *Crop Sci.* 39:1552–59
26. Peng S, Khush GS. 2003. Four decades of breeding for increased yield potential of irrigated rice in the International Rice Research Institute. *Plant Prod. Sci. (Jpn.)* 6(3):157–64
27. Peng S, Laza RC, Visperas RM, Sanico AL, Cassman KG, Khush GS. 2000. Grain yield of rice cultivars and lines developed in the Philippines since 1966. *Crop Sci.* 40:307–14
28. De Datta SK, Tauro AC, Baloing SN. 1968. Effect of plant type and nitrogen level on growth characteristics and grain yield of indica rice in the tropics. *Agron. J.* 60:643–47
29. Duvick DN, Cassman KG. 1999. Post-green revolution trends in yield potential of temperate maize in the North-Central United States. *Crop Sci.* 39:1622–30
30. Natl. Corn Grow. Assoc. 2003. *National Corn Growers Association—Corn Yield Contest*. http://www.ncga.com//02profits/CYC/main/index.html
31. Austin RB, Ford MA. 1989. Effects of nitrogen fertilizer on the performance of old and new varieties of winter wheat. *Vortr. Pflanzenzucht* 16:307–15
32. Sayre KD, Singh RP, Huerta-Espino J, Rajaram S. 1998. Genetic progress in

reducing yield losses to leaf rust in CIMMYT-derived Mexican spring wheat cultivars. *Crop Sci.* 38:654–59

33. Brancourt-Hulmel M, Doussinault G, Lecomte C, Berard P, Le Buanec B, Trotter M. 2003. Genetic improvement of agronomic traits of winter wheat cultivars released in France from 1946 to 1992. *Crop Sci.* 43:37–45
34. Matthews RB, Kropff MJ, Bachelet D, van Laar HH. 1995. *Modeling the impact of climate change on rice production in Asia.* Wallingford, UK: CAB Int., Int. Rice Res. Inst.
35. Sinclair TR, Horie T. 1989. Leaf nitrogen, photosynthesis, and crop radiation use efficiency: a review. *Crop Sci.* 29:90–98
36. Pretty J, Brett C, Gee D, Hine RE, Mason CF, et al. 2000. An assessment of the total external costs of UK agriculture. *Agric. Syst.* 65:113–36
37. Schweigert P, van der Ploeg RR. 2000. Nitrogen use efficiency in German agriculture since 1950: facts and evaluation. *Ber. Landwirtsch.* 80:185–212
38. Cassman KG, Dobermann A, Walters DT. 2002. Agroecosystems, nitrogen-use efficiency, and nitrogen management. *Ambio* 31:132–40
39. Poudel DD, Horwath WR, Lanini WT, Temple SR, van Bruggen AHC. 2002. Comparison of soil N availability and leaching potential, crop yields and weeds in organic, low-input and conventional farming systems in northern California. *Agric. Ecosyst. Environ.* 90:125–37
40. Di HJ, Cameron KC. 2002. Nitrate leaching in temperate agroecosystems: sources, factors and mitigating strategies. *Nutr. Cycl. Agroecosyst.* 64:237–56
41. Hansen B, Alroe HF, Kristensen ES. 2001. Approaches to assess the environmental impact of organic farming with particular regard to Denmark. *Agric. Ecosyst. Environ.* 83:11–26
42. Stivers LJ, Shennan C. 1991. Meeting the nitrogen needs of processing tomatoes through winter cover cropping. *J. Prod. Agric.* 4:330–35
43. Schluter W, Hennig A, Brummer G. 1997. Nitrate transport in soils of a river flood plain under organic and conventional farming - analytical results, modeling and balances. *Z. Pflanzenernaehr. Bodenk.* 160:57–65
44. Eltun R. 1995. Comparisons of nitrogen leaching in ecological and conventional cropping systems. *Biol. Agric. Hortic.* 11:103–14
45. Stopes C, Lord EI, Philipps L, Woodward L. 2002. Nitrate leaching from organic farms and conventional farms following best practice. *Soil Use Manag.* 18:256–63
46. Kirchmann H, Bergstroem L. 2001. Do organic farming practices reduce nitrate leaching? *Commun. Soil Sci. Plant Anal.* 32:997–1028
47. Flessa H, Wild U, Klemisch M, Pfadenhauer J. 1998. Nitrous oxide and methane fluxes from organic soils under agriculture. *Eur. J. Soil Sci.* 49:327–35
48. Flessa H, Ruser R, Dorsch P, Kamp T, Jimenez MA, et al. 2002. Integrated evaluation of greenhouse gas emissions ($CO_2$, $CH_4$, $N_2O$) from two farming systems in southern Germany. *Agric. Ecosyst. Environ.* 91:175–89
49. van der Werden TJ, Sherlock RR, Williams PH, Cameron K. 2000. Effect of three contrasting onion (*Allium cepa L.*) production systems on nitrous oxide emissions from soil. *Biol. Fertil. Soils* 31:334–42
50. Wang ZY, Xu YC, Li Z, Guo YX, Wassmann R, et al. 2000. A four-year record of methane emissions from irrigated rice fields in the Beijing region of China. *Nutr. Cycl. Agroecosyst.* 58:55–63
51. Wassmann R, Neue HU, Alberto MCR, Lantin RS, Bueno C, et al. 1996. Fluxes and pools of methane in wetland rice soils with varying organic inputs. *Environ. Monit. Assess.* 42:163–73
52. Mäder P, Fliessbach A, Dubois D, Gunst L, Fried P, Niggli U. 2002. Soil fertility

and biodiversity in organic farming. *Science* 296:1694–97

53. Eltun R, Korsaeth A, Nordheim O. 2002. A comparison of environmental, soil fertility, yield, and economical effects in six cropping systems based on an 8-year experiment in Norway. *Agric. Ecosyst. Environ.* 90:155–68
54. Dawe D. 2000. The contribution of rice research to poverty alleviation. In *Redesigning Rice Photosynthesis to Increase Yield,* ed. JE Sheehy, PL Mitchell, B Hardy, pp. 3–12. Makati City, Philipp./Amsterdam: Int. Rice Res. Inst., Elsevier Sci.
55. Senauer B, Sur M. 2001. Ending global hunger in the 21st century: projections of the number of food insecure people. *Rev. Agric. Econ.* 23:68–81
56. Dawe D, Dobermann A. 1999. *Defining productivity and yield. IRRI Discuss. Pap. Ser. 33.* Int. Rice Res. Inst. Makati City, Philipp.
57. Novoa R, Loomis RS. 1981. Nitrogen and plant production. *Plant Soil* 58:177–204
58. Cassman KG, Gines HC, Dizon M, Samson MI, Alcantara JM. 1996. Nitrogen-use efficiency in tropical lowland rice systems: contributions from indigenous and applied nitrogen. *Field Crops Res.* 47:1–12
59. Bell MA. 1993. Organic matter, soil properties, and wheat production in the high valley of Mexico. *Soil Sci.* 156:86–93
60. Kolberg RL, Westfall DG, Peterson GA. 1999. Influence of cropping intensity and nitrogen fertilizer rates on in situ nitrogen mineralization. *Soil Sci. Soc. Am. J.* 63:129–34
61. Cassman KG, Dobermann A, Santa Cruz PC, Gines HC, Samson MI, et al. 1996. Soil organic matter and the indigenous nitrogen supply of intensive irrigated rice systems in the tropics. *Plant Soil* 182:267–78
62. Dobermann A, Witt C, Abdulrachman S, Gines HC, Nagarajan R, et al. 2003. Estimating indigenous nutrient supplies for site-specific nutrient management in irrigated rice. *Agron. J.* 95:924–35
63. Tilman D, Fargione J, Wolff B, D'Antonio C, Dobson A, et al. 2001. Forecasting agriculturally driven global environmental change. *Science* 292:281–84
64. Int. Fertil. Ind. Assoc. 2002. *Fertilizer use by crop*. Rome: IFA, IFDC, IPI, PPI, FAO
65. Int. Fertil. Ind. Assoc. 2003. *IFADATA Statistics*. http://www.fertilizer.org/ifa/statistics.asp
66. Cassman KG, Pingali PL. 1995. Intensification of irrigated rice systems: learning from the past to meet future challenges. *GeoJ.* 35:299–305
67. US Dep. Agric., Natl. Agric. Stat. Serv. 2003. *Agricultural Statistics Database*. http://www.nass.usda.gov
68. Dobermann A, Witt C, Dawe D, Gines GC, Nagarajan R, et al. 2002. Site-specific nutrient management for intensive rice cropping systems in Asia. *Field Crops Res.* 74:37–66
69. Dobermann A. 2000. Future intensification of irrigated rice systems. In *Redesigning Rice Photosynthesis to Increase Yield*, ed. JE Sheehy, PL Mitchell, B Hardy, pp. 229–47. Makati City, Philipp./Amsterdam: Int. Rice Res. Inst., Elsevier Sci.
70. UN. 1998. *World Population Prospects.* New York: UN
71. Food Agric. Organ. UN. 2000. *Fertilizer Requirements in 2015 and 2030.* Rome: FAO UN
72. Bumb B, Baanante CA. 1996. *The role of fertilizer in sustaining food security and protecting the environment to 2020. Discuss. Pap. 17.* Washington, DC: Int. Food Policy Res. Inst.
73. Frink CR, Waggoner PE, Ausubel JH. 1999. Nitrogen fertilizer: retrospect and prospect. *Proc. Natl. Acad. Sci. USA* 96:1175–80
74. Dobermann A, Cassman KG. 2002. Plant nutrient management for enhanced productivity in intensive grain production systems of the United States and Asia. *Plant Soil* 247:153–75

75. Suzuki A. 1997. *Fertilization of Rice in Japan.* Tokyo, Jpn.: Jpn. FAO Assoc. 1–111 pp.
76. Mishima S. 2001. Recent trend of nitrogen flow associated with agricultural production in Japan. *Soil Sci. Plant Nutr.* 47:157–66
77. Peng S, Cassman KG. 1998. Upper thresholds of nitrogen uptake rates and associated N fertilizer efficiencies in irrigated rice. *Agron. J.* 90:178–85
78. Peng S, Garcia FV, Laza RC, Sanico AL, Visperas RM, Cassman KG. 1996. Increased N-use efficiency using a chlorophyll meter on high-yielding irrigated rice. *Field Crops Res.* 47:243–52
79. Dobermann A, White PF. 1999. Strategies for nutrient management in irrigated and rainfed lowland rice systems. *Nutr. Cycl. Agroecosyst.* 53:1–18
80. Witt C, Balasubramaniam V, Dobermann A, Buresh RJ. 2002. Nutrient management. In *Rice: A Practical Guide to Nutrient Management,* ed. TH Fairhurst, C Witt, pp. 1–45. Manila/Singapore: Int. Rice Res. Inst., Potash & Phosphate Inst./ Potash & Phosphate Inst. Can.
81. Wang GH, Dobermann A, Witt C, Sun QZ, Fu RX. 2001. Performance of site-specific nutrient management for irrigated rice in southeast China. *Agron. J.* 93:869–78
82. Matson PA, Naylor RL, Ortiz-Monasterio I. 1998. Integration of environmental, agronomic, and economic aspects of fertilizer management. *Science* 280:112–15
83. Riley WJ, Ortiz-Monasterio I, Matson PA. 2003. Nitrogen leaching and soil nitrate, nitrite, and ammonium levels under irrigated wheat in Northern Mexico. *Nutr. Cycl. Agroecosyst.* 61:223–36
84. Melillo JM, Prentice IC, Farquhar GD, Schulze ED, Sala OE. 1996. Terrestrial biotic responses to environmental change and feedbacks to climate. In *Climate Change 1995. The Science of Climate Change*, ed. JT Houghton, LG Meira Filho, BA Callander, N Harris, A Kattenberg, K Maskell, pp. 445–481. Cambridge, UK: IPCC, Cambridge Univ. Press
85. Metting FB, Smith JL, Amthor JS. 1998. Science needs and new technologies for soil carbon sequestration. In Carbon sequestration in soils: science, monitoring, and beyond. *Proc. St. Michaels Workshop, Dec.*, ed. NJ Rosenberg, pp. 1–34. Columbus, OH: Battelle
86. Intergov. Panel Clim. Chang. 1996. *Climate change 1995. The science of climate change*. Work. Group 1. Cambridge: IPCC., Cambridge Univ. Press.
87. Schlesinger WH. 1984. Soil organic matter: a source of atmospheric $CO_2$. In *The Role of Terrestrial Vegetation in the Global Carbon Cycle: Measurement by Remote Sensing,* ed. GM Woodwell, pp. 111–27. New York: Wiley
88. Houghton RA. 1995. Changes in the storage of terrestrial carbon since 1850. In *Soils and Global Change,* ed. R Lal, JM Kimble, E Levine, BA Stewart, pp. 45–65. Boca Raton, FL: CRC
89. West TO, Marland G. 2002. A synthesis of carbon sequestration, carbon emissions, and net carbon flux in agriculture: comparing tillage practices in the United States. *Agric. Ecosyst. Environ.* 91:217–32
90. Hair D, Sampson RN, Hamilton TE. 1996. Summary: forest management opportunities for increasing carbon storage. In *Forests and Global Change:* Vol. 2: *Forest Management Opportunities for Mitigating Carbon Emissions,* ed. RN Sampson, D Hair, pp. 237–54. Washington, DC: Am. Forests
91. Lal R, Kimble JM, Follett RF, Cole CV. 1998. *The Potential of U.S. Cropland to Sequester Carbon and Mitigate the Greenhouse Effect*. Chelsea, MI: Ann Arbor. 128 pp.
92. Intergov. Panel Clim. Chang. 2000. *Land use, land–use change, and forestry. Spec. Rep.* IPCC, Cambridge, UK
93. Paustian K, Andren O, Janzen HH, Lal R, Smith P, et al. 1997. Agricultural soils as

sink to mitigate $CO_2$ emissions. *Soil Use Manag.* 13:230–44

94. Dumanski J, Desjardins RL, Tarnocai C, Monreal C, Gregorich EG, et al. 1998. Possibilities for future carbon sequestration in Canadian agriculture in relation to land use changes. *Clim. Chang.* 40:81–103
95. Schlesinger WH, Lichter A. 2001. Limited carbon storage in soil and litter of experimental forest plots under increased atmospheric $CO_2$. *Nature* 411:466–69
96. Lal R, Kimble JM, Follett RF. 1999. Agricultural practices and policies for carbon sequestration in soil. *Recomm. Conclus. Int. Symp*. 19–23 July, Columbus, OH, 12 pp.
97. Waggoner PE. 1994. *How much land can ten billion people spare for nature? Task Force Rep. 121*. Counc. Agric. Sci. Technol. Ames, IA
98. Cassman KG. 2001. *Crop science research to assure food security*. In Crop science: progress and prospects. Presented at 3rd Int. Crop Sci. Congr., Hamburg, Ger., 2000, ed. J Nösberger, HH Geiger, PC Struik, pp. 33–51. Wallingford, UK: CABI
99. West TO, Post WM. 2002. Soil organic carbon sequestration rates by tillage and crop rotation: a global data analysis. *Soil Sci. Soc. Am. J.* 66:1930–46
100. Dick WA, Blevins RL, Frye WW, Peters SE, Christenson DR, et al. 1998. Impacts of agricultural management practices on C sequestration in forest-derived soils of the eastern Corn Belt. *Soil Tillage Res.* 47:235–44
101. Natl. Assess. Synth. Team. 2000. *Climate Change Impacts on the United States: The Potential Consequences of Climate Variability and Change.* Washington, DC: US Glob. Chang. Res. Program
102. Vleeshouwers LM, Verhagen A. 2002. Carbon emission and sequestration by agricultural land use: a model study for Europe. *Glob. Chang. Biol.* 8:519–30
103. Smith P, Goulding KWT, Smith KA, Powlson DS, Smith JU, et al. 2001. Enhancing the carbon sink in European agricultural soils: including trace gas fluxes in estimates of carbon mitigation potential. *Nutr. Cycl. Agroecosyst.* 60:237–52
104. Eve MD, Sperow M, Howerton K, Paustian K, Follett RF. 2002. Predicted impact of management changes on soil carbon storage for each cropland region of the conterminous United States. *J. Soil Water Conserv.* 57:196–204
105. Smith P, Powlson DS, Smith JU, Falloon P, Coleman K. 2000. Meeting Europe's climate change commitments: quantitative estimates of the potential for carbon mitigation by agriculture. *Glob. Chang. Biol.* 6:525–39
106. Jones CA, Kiniry JR. 1986. *CERES-Maize: A Simulation Model of Maize Growth and Development.* College Station: Tex. A&M Univ. Press
107. Muchow RC, Sinclair TR, Bennett JM. 1990. Temperature and solar radiation effects on potential maize yields across locations. *Agron. J.* 82:338–42
108. Kropff MJ, van Laar HH. 1993. *Modelling Crop-Weed Interactions*. Wallingford, UK: CABI
109. Parton WJ, Schimel DS, Cole CV, Ojima DS. 1987. Analysis of factors controlling soil organic matter levels in Great Plains grasslands. *Soil Sci. Soc. Am. J.* 51:1173–79
110. Metherell AK, Harding LA, Cole CV, Parton WJ. 1993. *Century Soil Organic Matter Model Environment. Technical Documentation, Agroecosystem Version 4.0.* Fort Collins, CO: USDA, Agric. Res. Serv.
111. Greenland DJ. 1998. Carbon sequestration in soil: knowledge gaps indicated by the symposium presentations. See Ref. 138, pp. 591–94
112. Izaurralde RC, Rosenberg NJ, Lal R. 2001. Mitigation of climatic change by soil carbon sequestration: issues of science, monitoring, and degraded lands. *Adv. Agron.* 70:1–75

113. Herrick JE, Wander MM. 1997. Relationship between soil organic carbon and soil quality in cropped and rangeland soils: the importance of distribution, composition and soil biological activity. See Ref. 138, pp. 405–25
114. Paustian K, Collins HP, Paul EA. 1997. Management controls on soil carbon. In *Soil Organic Matter in Temperate Agroecosystems*, ed. EA Paul, K Paustian, ET Elliott, CV Cole, pp. 15–49. Boca Raton, FL: CRC
115. Halvorson AD, Reule CA, Follett RF. 1999. Nitrogen fertilization effects on soil carbon and nitrogen in a dryland cropping system. *Soil Sci. Soc. Am. J.* 63:912–17
116. Bouwman AF. 1990. Exchange of greenhouse gases between terrestrial ecosystems and the atmosphere. In *Soils and the Greenhouse Effect*, ed. AF Bouwman, pp. 61–127. Chichester, UK: Wiley
117. Intergov. Panel Clim. Chang. 1996. *The Revised 1996 Guidelines for National Greenhouse Gas Inventories*. Vol. 1–3. Geneva, Switz.: IPCC
118. Smith KA, Mctaggart IP, Tsuruta H. 1997. Emissions of $N_2O$ and NO associated with nitrogen fertilization in intensive agriculture, and the potential for mitigation. *Soil Use Manag.* 13:296–304
119. Breitenbeck GA, Bremner JM. 1986. Effects of rate and depth of fertilizer application on emission of nitrous oxide from soil fertilized with anhydrous ammonia. *Biol. Fertil. Soils* 2:201–4
120. Simojoki A, Jaakkola A. 2000. Effect of nitrogen fertilization, cropping and irrigation on soil air composition and nitrous oxide emission in a loamy clay. *Eur. J. Soil Sci.* 51:413–24
121. Eichner MJ. 1990. Nitrous oxide emissions from fertilized soils: summary of available data. *J. Environ. Qual.* 19:272–80
122. Robertson GP, Paul EA, Harwood RR. 2000. Greenhouse gases in intensive agriculture: contributions of individual gases to the radiative forcing of the atmosphere. *Science* 289:1922–25
123. Collins HP, Blevins RL, Bundy LG, Christenson DR, Dick WA, et al. 1999. Soil carbon dynamics in corn-based agroecosystems: results from carbon-13 natural abundance. *Soil Sci. Soc. Am. J.* 63:584–91
124. Aulakh MS, Doran JW, Walters DT, Mosier AR, Francis DD. 1991. Crop residue type and placement effects of denitrification and mineralization. *Soil Sci. Soc. Am. J.* 55:1020–25
125. Weier KL, Doran JW, Power JF, Walters DT. 1993. Denitrification and the dinitrogen/nitrous oxide ratio as affected by soil water, available carbon, and nitrate. *Soil Sci. Soc. Am. J.* 57:66–72
126. Qian JH, Doran JW, Weier KL, Mosier AR, Peterson TA, Power JF. 1997. Soil denitrification and nitrous oxide losses under corn irrigated with high-nitrate groundwater. *J. Environ. Qual.* 26:348–60
127. Cates RL, Keeney DR. 1987. Nitrous oxide production throughout the year from fertilized and manured maize fields. *J. Environ. Qual.* 16:443–47
128. Cochran VL, Elliott LF, Papendick RI. 1981. Nitrous oxide emissions from a fallow field fertilized with anhydrous ammonia. *Soil Sci. Soc. Am. J.* 45:307–10
129. Bronson KF, Mosier AR, Bishnoi SR. 1992. Nitrous oxide emissions in irrigated corn as affected by nitrification-denitrification. *Soil Sci. Soc. Am. J.* 56:161–65
130. Papen H, Butterbach-Bahl K. 1999. A 3-year continuous record of nitrogen trace gas fluxes from untreated and limed soil of a N-saturated spruce and beech forest ecosystem in Germany. 1. $N_2O$ emissions. *J. Geophys. Res.* 104:18487–503
131. Hutchinson GL, Guenzi WD, Livingston GP. 1993. Soil water controls on aerobic emissions of gaseous N oxides. *Soil Biol. Biochem.* 25:1–9
132. Kessavalou A, Doran JW, Mosier AR, Drijber RA. 1998. Greenhouse gas fluxes

following tillage and wetting in a wheat-fallow cropping system. *J. Environ. Qual.* 27:1105–16

133. Hutchinson GL, Vigil MF, Doran JW, Kessavalou A. 1997. Coarse-scale soil-atmosphere $NO_x$ modeling: status and limitations. *Nutr. Cycl. Agroecosyst.* 48: 25–35

134. Schlesinger WH. 2000. Carbon sequestration in soils: some cautions amidst optimism. *Agric. Ecosyst. Environ.* 82:121–27

135. Pinstrup-Andersen P, Pandya-Lorch R, Rosegrant MW. 1999. *World Food Prospects: Critical Issues for the Early Twenty-First Century*. Washington, DC: Int. Food Policy Res. Inst. 32 pp.

136. Evans LT. 1998. *Feeding the Ten Billion: Plants and Population Growth*. Cambridge, UK: Cambridge Univ. Press

137. Dyson T. 1996. *Population and Food: Global Perspectives.* London: Routledge

138. Lal R, Kimble JM, Stewart BA, eds. 1998. *Soil Processes and the Carbon Cycle.* Boca Raton, FL: CRC