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# Agricultural Options for Mitigation of Greenhouse Gas Emissions

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# EXECUTIVE SUMMARY


Agriculture accounts for one-fifth of the annual increase in anthropogenic greenhouse warming. Most of this is due to methane ($CH_4$) and nitrous oxide ($N_2O$); agriculture produces about 50 and 70%, respectively, of their anthropogenic emissions. This chapter on mitigation options confirms earlier IPCC estimates and quantifies potential contributions by agriculture to reduce greenhouse gas emissions.

Between 400 and 800 Mt C/yr could be sequestered worldwide in agricultural soils by implementation of appropriate management practices to increase productivity—including increased input of crop residues, reduced tillage, and restoration of wasteland soils. However, soil carbon (C) sequestration has a finite capacity over a period of 50–100 years, as new equilibrium levels of soil organic matter are established.

Biofuel production on 10–15% of the land area currently in agricultural use or in agricultural set-asides could substitute for 300–1,300 Mt of fossil fuel C per year. Recovery and conversion of crop residues could substitute for an additional 100–200 Mt fossil fuel C per year. However, the possible offsets by increased $N_2O$ emissions need to be considered.

Energy use by agriculture relative to farm production has decreased greatly since the 1970s. Fossil fuel use by agriculture in industrialized countries, although constituting only 3–4% of overall consumption, can be further reduced by, for example, minimum tillage, irrigation scheduling, solar drying of crops, and improved fertilizer management.

Significant decreases in $CH_4$ emissions from agriculture can be achieved through improved nutrition of ruminant animals and better management of paddy rice fields. Additional $CH_4$ decreases are possible by altered treatment and management of animal wastes and by reduction of biomass burning. These combined practices could reduce $CH_4$ emissions from agriculture by 15–56%.

The primary sources of $N_2O$ are mineral fertilizers, legume cropping, and animal waste. Some $N_2O$ also is emitted from biomass burning. $N_2O$ emissions from agriculture could be reduced by 9–26% by improving agricultural management with available techniques.

Ranges in estimates of potential mitigation reflect uncertainty in the effectiveness of recommended management options and in the degree of future implementation globally. Comprehensive model-based analyses and improved global databases on soils, land use, and trace gas fluxes are needed to reduce the uncertainty in these estimates.

Farmers will not voluntarily adopt greenhouse gas (GHG) mitigation techniques unless they improve profitability. Some techniques, such as no-till agriculture or strategic fertilizer placement and timing, already are being adopted for reasons other than concern for climate change. Proposed options for reducing GHG emissions are consistent with maintaining or increasing agricultural productivity.

## 23.1. Introduction

Agriculture contributes significantly to anthropogenic emissions of carbon dioxide ($CO_2$), methane ($CH_4$), and nitrous oxide ($N_2O$). Based on IPCC data (IPCC, 1994) and additional estimates described below, agriculture accounts for about one-fifth of the annual increase in radiative forcing (Figure 23-1). Land-use changes related to agriculture especially in the tropics, including biomass burning and soil degradation, are also major contributors.

The large influence of agriculture needs to be traced back to its individual sources so that effective mitigations can be sought. Options such as those proposed by the International Workshop on Greenhouse Gas Emissions from Agricultural Systems (U.S. EPA, 1990a), and others that have been proposed more recently, are shown in Table 23-1.

The IPCC First Assessment Report (IPCC, 1990a, 1990b) concluded that land-use changes are the most important source of anthropogenic $CO_2$ after fossil fuel combustion. Improving the productivity of existing farmland was proposed to mitigate these emissions. Regarding the other two trace gases, it was suggested that $CH_4$ from rice fields could be reduced by 20–40% and $CH_4$ from ruminant animals by 25–75%. Improving Nitrogen (N) fertilizer efficiency and minimizing N surpluses were cited as important approaches for reducing agricultural $N_2O$ emissions, but no quantitative assessment was made at that time.

Subsequent IPCC assessments (IPCC, 1991, 1992) corroborated most of the earlier suggestions, but estimates of $CH_4$ emissions from rice were considerably reduced, and animal manure was identified as an additional $CH_4$ source. The importance of $N_2O$ emissions from heavily fertilized soils was further emphasized, but the quantities involved still remained highly uncertain.

Since then, the availability of new regional and global databases (FAO, 1990a, 1990b, 1990c; Eswaran *et al*., 1993; Sombroek *et al*., 1993), additional site-specific measurements, and the utilization of more thoroughly tested simulation models (Raich *et al*., 1991; Ojima *et al*., 1993c; Parton *et al*., 1993; Potter *et al*., 1993) have improved the possibilities for quantification. However, information concerning greenhouse gas (GHG) fluxes from agricultural systems is still limited.

Global food production needs and farmer/society acceptability considerations suggest that mitigation technologies should meet the following general guidelines: (1) agricultural production levels will be maintained or enhanced in parts of the world where food production and population demand are in delicate balance; (2) additional benefits will accrue to the farmer (e.g., reduced labor, reduced or more efficient use of inputs); (3) agricultural products will be accepted by local consumers.

Table 23-1 summarizes greenhouse gas mitigation options related to agriculture. These options are grouped into different activities, and the list includes a qualitative assessment of their relevance for the three trace gases $CO_2$, $CH_4$, and $N_2O$. Emphasis in this chapter will be given to quantifying these mitigation potentials.

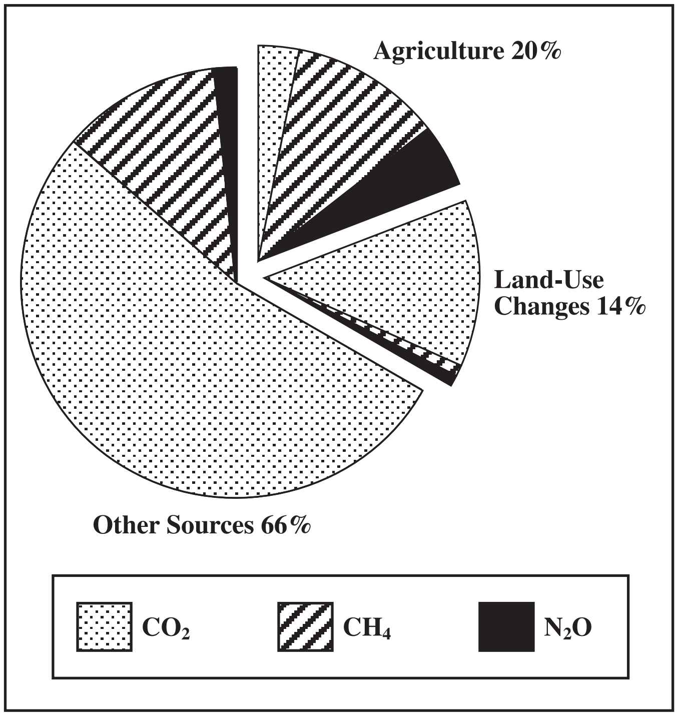


**Figure 23-1:** Proportions of the annual increase in global radiative forcing attributable to agriculture and agriculture-related land-use changes.

## 23.2. Carbon Dioxide

Options to mitigate $CO_2$ emissions from agriculture and land-use changes include the reduction of emissions from present sources and the creation and strengthening of carbon sinks. Increasing the role of agricultural land as a sink for $CO_2$ includes C storage in managed soils and C sequestration after reversion of surplus farm lands to natural ecosystems (Follett, 1993). Biofuel production on agricultural lands has a considerable potential for mitigation of $CO_2$ emissions by providing C offsets to use of fossil fuel.

### *23.2.1. Land-Use Changes*

#### *23.2.1.1. Land-Use Changes in the Tropics*

Biomass burning and loss of soil C associated with the conversion of native ecosystems to agricultural use in the tropics is believed to be the largest non-fossil fuel source of $CO_2$ input to the atmosphere. The net release of $CO_2$ from land-use conversion is thought to be in the range of 1.6 ± 1.0 Gt C/yr (IPCC, 1994). Of the C losses attributed to land use, soil C loss has been estimated to account for 20–40% (Detwiler, 1986;

Houghton and Skole, 1990). Recent data, however, suggest that soil C losses following deforestation may have been overestimated, particularly for forest conversions to pasture, where soil C can recover to levels equal to or higher than native forest within a few years (Lugo and Brown, 1993; Cerri *et al*., 1994). However, a reduction in land conversion rates remains an important $CO_2$ mitigation option (see Chapter 19).

Population increase, unresolved land tenure issues, desire for higher living standards, and other sociopolitical factors drive the demand for new cropland in the tropics. However, the suitability and management of this new cropland is often poor, resulting in soil degradation that exerts additional pressure to convert new lands to agriculture. Currently only half of the conversion of tropical forests to agriculture contributes to an increase in productive agricultural area; the other half is used to replace previously cultivated land that has been degraded and abandoned from production (Houghton, 1994). The only way to break out of this cycle is through more sustainable use and improved productivity of existing farmland (Sanchez *et al*., 1990) and better protection of native ecosystems.

The mitigation options associated with land-use changes are strongly related to major climatic zones. The most significant opportunities appear to be in the humid tropics and in tropical wetlands. In sub-humid zones, much of the land area has already been converted to permanent agriculture. However, improved management and productivity on these lands could help to reduce agricultural expansion (and hence deforestation) in humid zones, especially in Latin America and Africa. In the

***Table 23-1:*** *Options for direct and indirect mitigation of greenhouse gas emissions from agriculture.*

| | $CO_2$ | $CH_4$ | $N_2O$ |
|---|---|---|---|
| **1. Land Conversion and Management** | | | |
| – Reduced deforestation rate | H | M | M |
| – Pasture immediately after deforestation | M | | L |
| – Conversion of marginal agricultural land to grassland, forest, or wetland | M | | L |
| **2. Agricultural Land Utilization and Management** | | | |
| – Restoring productivity of degraded soils | H | L | |
| – More intensive use of existing farmland | M | L | L |
| – Restrict use of organic soils | H | | L |
| – Conservation tillage | M | | L |
| – Reduction of dryland fallowing | M | | |
| – Diversified rotations with forage crops | M | | L |
| **3. Biofuels** | | | |
| – Energy crops for fossil fuel substitution | H | | |
| – Agroforestry | L | | |
| – Windbreaks and shelterbelts | L | | |
| – Agroindustrial wastes for fossil fuel substitution | L | | |
| **4. Recycling of Livestock and Other Wastes** | | | |
| – Recycling of municipal organic wastes | L | L | M |
| – Biogas use from liquid manures | | M | |
| **5. Animal Husbandry** | | | |
| – Supplementing low-quality feed | | M | |
| – Increasing feed digestibility | | L | |
| – Production-enhancing agents | | L | |
| **6. Rice Cropping Systems** | | | |
| – Irrigation management | | M | L |
| – Nutrient management | | M | L |
| – New cultivars and other | | M | L |
| **7. Plant Nutrient Management** | | | |
| – Improved fertilizer use efficiency | L | | H |
| – Nitrification inhibitors | | | M |
| – Legume cropping to bolster system productivity | | | M |
| – Integrating crop and animal farming | | | L |
| **8. Minimizing Overall N Inputs** | | | |
| – Reduced protein inputs in animal feed | | | M |
| – Reduced protein consumption by society | | M | M |

Note: Mitigation Potential: L = Low, M = Medium, H = High.

semi-arid zone, soil and biomass C stocks are smaller and differ less as a function of land use; therefore, the scope for $CO_2$ mitigation through changing land-use patterns is more limited.

*23.2.1.2. Reversion of Agricultural Land in Temperate Zones*

In temperate regions there is little development of new agricultural lands, and in regions with food surpluses (e.g., United States, Canada, Western Europe) the agricultural land base is being reduced. A similar situation may occur in the longer term for countries in Eastern Europe and the former Soviet Union (FSU) as per area productivity increases. Thus, the reversion of marginal agricultural land to forest (including shelterbelts and plantations), grassland, and wetlands represents a potential for C sequestration.

Rates of C accumulation in reverted agricultural soils vary greatly depending on climate and soil conditions, the vegetation type established, and the degree of management (Table 23-2).

Upon release from agricultural production, C sequestration would continue only until soils reached a new equilibrium value, most of which would be realized over a 50–100 year period. An exception is the case of reversion to wetlands, where the buildup of organic soils provides a more sustained C sink. About 8 Mha of organic soil-wetlands are currently under cultivation in the temperate zone. As of 1980, these soils were estimated to provide a net C source of ~5 Mt/yr, but represent a potential sink of ~55 Mt C/yr in their native state, with sustained rates of C accumulation of 20–225 g C/m$^2$/yr (Amentano and Menges, 1986).

Currently about 25 Mha in the United States, Canada, and the European Union (EU) (~15% of total cropland) has been taken out of production by government set-aside programs. The U.S. Conservation Reserve Program, started in 1985, is scheduled to begin expiring in 1996, and 75% of the land is expected to return to agriculture by 2005 unless the program is extended. The EU agricultural set-aside programs are for 1–5 years and can include annual cropping for non-food production (e.g., oilseed for fuel), although separate programs exist for tree planting. Because C stocks are likely to be depleted again when lands are returned to cultivation, short duration set-asides have little or no effect on C sequestration. If soils are left uncultivated and allowed to revert to native vegetation, C contents in upper soil horizons could eventually reach levels comparable to their precultivation condition. Considering the ~640 Mha of cropland with real or potential surpluses (United States, Canada, FSU, Europe, Australia, and Argentina) and assuming recovery of the soil C originally lost due to cultivation (25–30% per Davidson and Ackerman, 1993), a permanent set-aside of 15% of the land area might sequester 1.5–3 Gt C.

A large-scale reversion or afforestation of agricultural land is possible only if adequate supplies of food, fiber, and energy can be obtained from the remaining area. This is currently possible in the EU and United States through intensive farming systems. However, if farming intensity changes because of environmental concerns or changes in policy (Enquete Commission, 1995; Sauerbeck, 1994a), this mitigation option may no longer be available.

### 23.2.2. Carbon Sequestration in Agroecosystems

Losses of soil C as a consequence of cultivation are ubiquitous and well-documented (Haas *et al.*, 1957; Greenland and Nye, 1959; Davidson and Ackerman, 1993). Historical losses of C observed in many soils were due, in part, to low production levels, erosion, inadequate fertilization, removal of crop residues, and intensive tillage. Improved management is capable of

***Table 23-2:*** *Examples of C sequestration rates with conversion of agricultural land to forest and grasslands in some temperate locations.*

| Region | Land-Use Conversion | Time Period (yr) | Soil Depth Considered (cm) | Mean Soil C Accumulation (g/m$^2$/yr) |
|---|---|---|---|---|
| UK | Abandonment to deciduous forest[a,b] | ~100 | 23 | 50 |
| UK | Planted grassland[c] | 15 | 15 | 75 |
| Canada | Abandonment to native shortgrass prairie[d] | 24 | 15 | 13 |
| USA | Grassland seeded for set-aside program[e] | 5 | 100 | 15–120 |
| USA | Abandonment to native shortgrass prairie[f] | 50 | 10 | 3 |
| New Zealand | Planted grassland[g] | 18 | 20 | 100 |

[a]Jenkinson, 1971.
[b]Jenkinson *et al.*, 1992.
[c]Tyson *et al.*, 1990.
[d]Dormaar and Smoliak, 1985.
[e]Gebhart *et al.*, in press.
[f]Burke *et al.*, submitted.
[g]Haynes *et al.*, 1991.

***Table 23-3:*** *Soil C contents (1 m) and historic soil C loss from cultivated soils worldwide.*

| **Soil Group**[a] | **Cultivated Area (Mha)** | **C Mass—Virgin**[b] **(Gt C)** | **C Mass—Cultivated (Gt C)** | **Soil C Loss**[c] **(Gt C)** |
|---|---|---|---|---|
| Forest Soils | 822 | 85 | 63 | 22 |
| Grassland Soils | 438 | 58 | 43 | 15 |
| Wetland Soils | 128 | 54 | 43 | 11 |
| Volcanic Soils | 31 | 7 | 5 | 2 |
| Other Soils | 308 | 18 | 13 | 5 |
| **Total** | **1727** | **222** | **167** | **55** |

[a] Soil groups are aggregated according to the global distribution of cultivated soils in major FAO soil groups by Bouwman (1990). Aggregations included the following soil types: Forest (Acrisols, Podzoluvisols, Ferralsols, Luvisol, Cambisol, Nitosol, Podzol, Chromic Luvisol, Planosol), grassland (Chernozem, Phaeozem, Greyzem, Kastanozem, Vertisol), wetland (Gleysol, Histosol), volcanic (Andosol), and other (Lithosol, Arenosol, Regosol, Solonchak, Solonetz, Xerosol). Some soil types (e.g., Acrisols) that contain subgroups occurring in different vegetation types (e.g., forest vs. grassland) were aggregated under a single soil group (e.g., forest) designation.

[b] C contents, to 1m depth, were calculated separately for each FAO soil type (excluding Histosols) using data from Sombroek *et al*. (1993), then aggregated. C contents for Histosols (112 kg/$m^2$) were from Amentano and Menges (1986).

[c] Carbon loss due to cultivation of mineral soils was assumed to be 26% of C in the top 30 cm, based on the average reported by Davidson and Ackerman (1993). Losses were calculated separately for the surface layer of soil groups given by Sombroek *et al*. (1993), then aggregated. Estimates of C lost from cultivation of Histosols to 1990 (8 Gt C) were from Amentano and Menges (1986). C mass in cultivated soils was calculated as the difference between uncultivated C contents and C loss estimates.

increasing C levels on present agricultural soils and reducing losses from newly cultivated soils. In general, high residue production, perennial forage crops, elimination of bare fallow, and reduced tillage will promote C sequestration.

An estimate based on the distribution of cultivated soils across major soil groups, their associated C contents, and average loss rates due to cultivation yields a current global stock in cultivated lands of 167 Gt C and a historical loss from these soils of 55 Gt C (Table 23-3). Comparable estimates have been obtained by Houghton and coworkers (Houghton *et al*., 1983; Houghton and Skole, 1990) from a more detailed analysis of historical land-use data. They estimated global losses of soil C since 1860 to be 30 Gt C (15% of the total 170 Gt C lost from vegetation and soils), or 41 Gt C since the beginning of settled agriculture (Houghton and Skole, 1990). These land use-based estimates do not include C loss from drainage of wetland soils, which could account for the higher value calculated in Table 23-3.

These global estimates of C loss from cultivated soils provide a reference level for the C sequestration that might be achieved through improved management. Increasing C levels in artificially drained wetland soils (e.g., Histosols) is unlikely except in cases where they are reverted to wetlands. Therefore, the potential to increase C levels in cultivated systems is largely restricted to upland soils. Assuming a recovery of one-half to two-thirds of historic C losses as a reasonable upper limit, the global potential for C sequestration in cultivated soils over the next 50–100 years would be on the order of 20–30 Gt. This would require increased production and major improvements in management on much of the world's cultivated areas, particularly in less-developed regions. Under current conditions, it is estimated that there is little or no net C sequestration in temperate soils as a whole, and cultivated soils in the tropics are probably a net source of C (Cole *et al*., 1993; Sauerbeck, 1993). To identify the most important management factors and constraints on C sequestration, more detailed regional information is provided in Sections 23.2.2.1 through 23.2.2.3.

#### *23.2.2.1. Tropical Agroecosystems*

The decrease in soil C with permanent cultivation is of the same order as that for temperate regions, often 20–50% or more, although C losses typically are more rapid in the tropics. It is estimated that arable land and pasture use in the tropics currently contributes a net C flux of 90–230 Mt C annually (Lal and Logan, in press). Decreases in soil C are a result of lower organic matter inputs relative to native systems, enhanced decomposition rates, and/or erosion. Management to increase soil C levels is a high priority for improving the productivity and sustainability of tropical agricultural systems (Swift and Sanchez, 1984). Opportunities to increase soil C levels include improved management of presently cultivated land and the restoration of degraded lands. The latter will include conversion to nonagricultural uses such as forest or biofuel production, but some areas may be suitable for agricultural production as well.

Management practices to increase soil C stocks include reduced tillage, increased production and crop residue return, perennial crops (including agroforestry), and reduced bare fallow frequency. Introduction of deep-rooting cultivars of tropical grasses also shows promise (Fisher *et al*., 1994). However, there are economic, educational, and sociological constraints to improved soil management in much of the tropics. Many tropical farmers cannot afford or have limited access to purchased inputs such as fertilizer and herbicides. Crop residues often are needed for livestock feed, fuel, or other household uses, which reduces C inputs to soil (Elwell, 1993). On the other hand, development efforts that seek to increase the sustainability and productivity of tropical agriculture will be largely compatible with $CO_2$ mitigation needs. To the extent that improved management is based on significantly increased fossil fuel consumption, however, benefits for $CO_2$ mitigation will be decreased.

Increasing soil C levels depends on increasing crop productivity and input of organic matter to the soil (through irrigation, soil fertility improvement, multiple-cropping, controlled grazing) and/or decreasing decomposition rates (through reducing tillage, residue quality, pH, temperature, moisture). The relative importance of these factors will vary according to climate and soil type.

In much of the semi-arid tropics, the predominant land use is pastoral, and grazing control is an important option for maintenance of soil C. Data from northeastern Brazil suggest that improved pastures can maintain C levels comparable to those of native systems (1–1.5% C), which are roughly twice those in shifting cultivation systems (Tiessen, pers. comm.). Water availability is the main determinant of productivity, and the use of reduced tillage and mulching to increase available water and to reduce surface erosion can promote increased soil C (Lal, 1986). In areas where cropping and livestock are closely integrated, such as parts of the west African Sahel and the Miombo zone in southern Africa, better efficiency of manure utilization and supplemental commercial fertilizer can increase productivity and soil C (Feller and Garin, pers. comm.). Overall, however, C stocks are low even under native ecosystems, and therefore $CO_2$ mitigation options are limited.

In sub-humid and humid zones, $CO_2$ mitigation potentials are higher. With greater water availability, the physical condition and fertility of soils become more important as production constraints. The inherent physical conditions of these soils are generally good but can be degraded with intensive tillage. Reduced tillage, mulch farming, alley cropping, and other agroforestry practices can contribute to maintenance of soil structure and reduction of erosion and can provide increased organic matter inputs and reduced decomposition rates (by reducing soil temperatures and soil disturbance). In wetland soils, especially paddy fields, low aeration slows decomposition and leads to high C contents, especially in continuously wet soils. When soils dry up during part of the cropping cycle, C contents will decrease unless green manures or large amounts of crop residues (e.g., wheat) are produced in the dry season. The widespread burning of rice straw reduces organic C inputs to the soil. Higher C stocks in wetland soils generally increase $CH_4$ emissions, although better water management can help control $CH_4$ emissions.

Restoration of degraded lands is a significant mitigation option. Worldwide there are about 1,200 Mha of moderately to severely degraded lands, which constitute a large potential C sink (Oldeman *et al.*, 1990). In the tropical and subtropical regions of China, about 48 Mha is classified as "wasteland," of which 80% is considered suitable for forestry and 8% and 6% could be restored as cropland and pasture, respectively (Zhao, pers. comm.). Based on field experiments, net C increases in soils reclaimed for agriculture are on the order of 2.4 kg C/m$^2$ over a 10-year period, which represents a net C sink of 92 Mt C. Higher increases could be obtained for the larger area suitable for reforestation. In India, more than 100 Mha are classified as degraded and greatly depleted in soil C. Experiments have shown that salt- and alkali-affected soils have a relatively high potential for C sequestration if suitable tree and grass species and water management measures are used (Gupta and Rao, 1994). These authors estimated that a restoration of 35 Mha of wasteland area in India could sequester up to 2 Gt C.

#### *23.2.2.2. Temperate Agroecosystems*

Most temperate agricultural soils have been cultivated for decades to centuries, and most probably have attained a C content close to equilibrium value. Very little land in the temperate zone is currently being converted to agriculture. For industrialized agriculture (i.e., United States, EU), soil C probably is increasing slightly due to increased productivity and hence greater crop residue inputs, improved residue management, and reduced tillage. In less industrialized agriculture (e.g., FSU) and in areas that have been brought into cultivation more recently, soil C probably is decreasing slightly. As a first approximation, it is reasonable to assume that temperate agricultural soils are not a large source or sink of C under current practices (Cole *et al.*, 1993; Sauerbeck, 1993). However, soil organic C in permanently cropped fields can be increased through a number of management practices, including greater returns of organic materials to soil, decreased periods of fallow, use of perennial and winter cover crops, recycling of organic wastes, reduced tillage, erosion control, and agroforestry.

Soil C levels are closely tied to the rate of C return from crop residues and other sources. Numerous long-term field experiments demonstrate that for many soils, organic C levels are directly proportional to the annual rate of C input (Rasmussen and Collins, 1991; Paustian *et al.*, 1995). Increasing crop production through better nutrient management, reduced fallow periods, and improved cultivars can increase C inputs to soil if crop residues are retained.

Summer fallow is used extensively in semi-arid areas of Canada, the United States, Australia, and the FSU to offset rainfall variability and increase soil water storage. Eliminating or reducing summer fallow through better water management could significantly increase C in semi-arid croplands and decrease soil erosion (Janzen, 1987; Campbell *et al.*, 1990). During fallow periods, mineralization of soil organic C generally is faster than under a crop, and there is no input of crop residues (e.g., in a spring wheat-fallow rotation, there may be only 4 months of crop cover per 24 months).

Greater use of perennial forage crops can significantly increase soil C levels, due to high root C production, lack of tillage disturbance, and protection from erosion. Increases of up to 100 g C/m$^2$/yr have been documented for cultivated land planted to grassland (see Table 23-2). Where climate permits, winter cover crops decrease erosion and provide additional inputs of C, thereby increasing soil organic carbon (SOC). Simulation of the U.S. cornbelt with the EPIC model suggested an additional C sequestration of 400 g C/m$^2$ in 100 years from the use of winter cover crops (Lee *et al*., 1993).

Large applications of manure can increase soil C as much as can reversion to natural vegetation. For example, in the Broadbalk Wheat Experiment at Rothamsted, UK, the application of farmyard manure (FYM) at the very high rate of 35 t/ha annually since 1843 has increased %C in topsoil (0–23 cm) from 0.92% (measured in 1881) to 2.8%. However, such an application rate is not possible on a large scale, and the off-site impacts of large manure application need to be considered.

Reduced- or no-till systems often (but not always) increase soil C. Reduced tillage generally causes organic matter to be concentrated near the surface, but this does not necessarily represent an increase within the total profile (Powlson *et al*., 1987). However, several studies have shown genuine increases in SOC as a result of reduced tillage (e.g., Dick *et al*., 1986; Saffigna *et al*., 1989; Balesdent *et al*., 1990; Ismail *et al*., 1994). For the United States, Kern and Johnson (1993) estimate that an increase in the use of reduced tillage from current levels of 27% to 76% of cultivated area by the year 2020 would result in a net C sequestration of 0.2–0.3 Gt C versus a net loss of 0.2 Gt C with current practices. Gaston *et al*. (1993) estimate that no-till management of 181 Mha of climatically suitable land in the FSU could sequester as much as 3.3 Gt C.

In the FSU, significant effort has been made to prevent losses of soil organic matter by using straw mulch and shelterbelts, and terracing steep slopes. Planting of 18 Mha of shelterbelts and afforestation of 53 Mha of degraded landscapes and heavily polluted lands would allow 16 Gt C to be sequestered and conserved over 100 years (160 Mt C/yr)(Kolchugina and Vinson, pers. comm.).

### *23.2.2.3. Regional Analyses*

A rigorous assessment of soil C changes and the potential impacts of various mitigation strategies requires the integration of information on land-use and management practices, soils, and climate at regional scales. The growing availability of national and, to a lesser extent, global databases, together with the use of simulation modeling, provide the framework for such analyses. While no such global-scale analyses of C in agricultural systems have yet been done, three regional analyses for areas in North America illustrate the utility of such an approach, as well as problems and information constraints.

Impacts of different management on soil C content for the U.S. cornbelt region over a 100-year period were assessed by Lee *et al*. (1993) using the EPIC model (Sharpley and Williams, 1990). This study suggested continued losses of C under the current mix of tillage practices. However, widespread use of no-till plus winter cover crops resulted in a net accumulation of 0.10 Mt C/yr. Lee *et al*. (1993) projected that the net effect of a shift to widespread use of no-till plus winter cover crops could conserve 3.3 Mt C/yr over the next 100 years.

The potential C sequestration in Canadian agricultural soils over the next 50 years was assessed by Dumanski *et al*. (pers. comm.) using long-term field data and the Century model (Parton *et al*., 1987). They assumed that cropland area would remain within 5% of the current area; that there would be a major reduction in summer/fallow in the Chernozemic soil zone; and that cropping practices would be intensified, with increased fertilizer use, improved residue management with reduced tillage, and better erosion control. Over a 50-year period they projected an increased storage of 22 Mt C due to reduction of summer fallow and 69 Mt C from increased hay crops in rotations. Proper fertilization and erosion control through zero-tillage and other measures were projected to further increase C. Summing these management treatments, C equivalent to 3.4% of Canada's present $CO_2$ emissions could be sequestered annually in improved croplands.

Potential impacts of alternative management practices on soil C in the major agricultural regions of the central United States were analyzed by Donigian *et al*. (1994). They conducted a detailed analysis combining geographic databases on climate, soils, and land use with Century model (Parton *et al*., 1987) simulations. Management scenarios included increased use of cover crops, increased adoption of reduced tillage, and impacts of land set-aside, as well as potential increases in crop production. This analysis projected an increase in net soil C, primarily due to increases in productivity and crop residue inputs. Thus, a continuation of current trends in agricultural practices would increase C storage by 25 to 50 Mt C per year in a region comprising 70% of U.S. cropland (87 Mha).

A global-level analysis using approaches similar to those outlined above is needed in order to better evaluate potential C changes, including an analysis of management effects and interactions with potential climate change. However, databases on land use and management, soils, and climate for many developing countries are not adequate to provide a reliable framework for analysis. Uncertainties about the distribution of land-use types and management regimes in relation to soils and climate and the behavior of soil organic matter below the surface horizon are important topics that need to be addressed. The current efforts of Food and Agriculture Organization (FAO) to improve such databases at both national and global levels should be expanded.

### 23.2.3. Fossil Energy Use by Agriculture

Fossil energy use by agriculture is about 3–4.5% of the total consumption in the developed countries of the world (CAST, 1992; Haas *et al*., 1995a, 1995b; Enquete Commission, 1995). Thus, for the primary farm production sector, the mitigation potentials through reduced fuel consumption are relatively small. The ratio of energy use to farm production has decreased markedly since the "oil crisis" of 1973 sharply increased the costs of energy to all sectors including agriculture (Darmstadter, 1993) and continues to decrease at this time (U.S. Government Executive Office, 1995). Further reductions may be expected by expanded use of minimum tillage, irrigation scheduling, solar drying of crops, and improved fertilizer management.

Tillage and harvest operations account for the greatest proportion of fuel consumption in intensive cropping systems. Considerable energy savings are possible because the fuel requirements using no-till or reduced tillage are 55% and 78%, respectively, of that for conventional moldboard plow tillage (Frye, 1984).

The fixation of atmospheric N into synthetic fertilizer requires about 1.2 kg of fossil C equivalents for each kg of fixed N. Therefore, the present global consumption of about 80 Mt fertilizer N corresponds to the consumption of 100 Mt fossil C per year. Although the N fertilizer use in developed countries may not increase much further, it is predicted to double in developing countries by the year 2025 (IPCC, 1992; Sauerbeck, 1994a). Accordingly, the energy required for manufacturing N fertilizers will increase to about 150 Mt fossil C annually. In Southeast Asia, for example, a 30% increase in the use of fossil fuel is considered necessary by the year 2000, if food production is to follow population and economic growth. Thus, optimizing N use efficiency and minimizing N surpluses provide mitigation of $CO_2$ emissions in addition to reducing $N_2O$ emissions.

High-intensity animal production has become the biggest consumer of fossil energy in modern agriculture (van Heerwarden *et al*., 1992; Enquete Commission, 1995). This needs to be considered more critically, as do recent emissions comparisons between conventional and alternative agricultural systems (Enquete Commission, 1995). As an example, Haas *et al*. (1995a, 1995b) calculated that organic farm systems in Germany emit only 39% of the overall fossil C required by conventional farms. This is due mainly to the replacement of mineral N fertilizers by legume cropping, balanced animal stocking rates, and a much lower consumption of feed concentrates (Koepf *et al*., 1988; Enquete Commission, 1995). Even energy inputs per ton of harvested crop were lower by 20–60% (Haas *et al*., 1995a, 1995b), although this depends on yield levels and cannot be generalized (Nguyen and Haynes, 1995).

Fuel requirements by the food sector as a whole (including processing, preservation, storage, and distribution) account for 10–20% of total fossil energy consumption (Pimentel *et al*., 1990; CAST, 1992; Haas *et al*., 1995a, 1995b). Thus, it may be worthwhile to reconsider the extent of food transport and the overall nutritional habits of the societies in the developed world. For instance, reducing animal protein consumption in Europe and the United States by only one-half of its present excess would decrease N fertilizer requirements by about one-half (Sauerbeck, 1994b; Isermann and Isermann, 1995).

### 23.2.4. Biofuel Production

Both non-food crops and many conventional agricultural crops produce biomass that is valuable as a feedstock for either energy supply or industrial products. Most of these crops require soil and management conditions similar to food crops and compete for limited resources. The extent to which their production will be expanded in the future depends on the development of new technologies, their economic competitiveness with traditional food and fiber crops as well as with conventional petrochemical feedstocks, and social and political pressures for more renewable and biodegradable materials.

The greatest agricultural potential for mitigating $CO_2$ lies in increasing the amount and variety of plant biomass used directly for energy production (herein defined as biofuels; see also Chapter 19) as a substitute for fossil energy. This increase could be realized by substituting biofuel crops for other agricultural crops (particularly those in surplus supply), by growing them on lands held in agricultural set-aside programs, or by intermixing biofuel plants with food or forage plants in an agroforestry system.

There are also significant opportunities to utilize crop residues and byproducts for the production of energy to replace fossil fuels. These vary widely, however, in terms of the feasibility of collecting and transporting residues, as well as the extent to which crop residues can be removed from fields without adversely affecting soil C levels and site productivity. In general, it is estimated that only 50% of the residues can be removed without affecting future soil productivity, and only 25% should be considered as recoverable for energy purposes (Sampson *et al*., 1993). Hall *et al*. (1993) estimate that for the world's major crops (wheat, rice, corn, barley, and sugarcane), a 25% residue recovery rate would amount to an annual C amount of 357 Mt. Much of this would be used for heating and cooking, without substituting for coal or petroleum use. Based on assumptions for energy conversion and degree of substitution for fossil C, crop residues could offset 100–200 Mt fossil C.

Other opportunities include converting marginal or surplus crop and pasture land to forest, increasing the use of forest biomass, noncommercial thinning of industrial timber and paper production wastes, and using recycled wood and paper products for biofuels. Many of the agricultural biofuels considered could be advantageously combined with forest biofuels to stagger harvest dates, reduce storage facility needs, create a more uniform year-round feedstock supply, and reduce the collection radius needed for a feasible utilization facility such as a utility-scale electrical power station.

The $CO_2$ mitigation potential of a large-scale global agricultural biofuel program can be significant (Johansson *et al*., 1993). Assuming that 10–15% of the world's cropland area could be made available, fossil fuel substitutions in the range of 0.3 to 1.3 Gt C/yr have been estimated. This estimate does not include the indirect $CO_2$ offsets of biofuel production through increasing C storage in standing woody biomass, and possibly by increased soil C sequestration.

#### *23.2.4.1. Dedicated Biofuel Crops*

Dedicated energy plants, including short-rotation woody crops (SRWC), perennial herbaceous energy crops (HEC), and annuals such as whole-plant cereal crops, could be sustainably grown on 8–11% of the marginal to good cropland in the temperate zone (Table 23-4; Sampson *et al*., 1993). For example, in the EU it has been estimated that 15–20 Mha of good agricultural land will be surplus to food production needs by the year 2010 (Scurlock *et al*., 1993; Flaig and Mohr, 1994). This would be equivalent to 20–30% of the current cropland area.

Due to increasing agricultural demand in the tropics, a lower percentage of land is likely to be dedicated to energy crops, so a reasonable estimate may be 5–7% (Sampson *et al*., 1993). In total, however, there could be a significant amount of land available for biofuel production (FAO, 1994), especially from marginal land and land in need of rehabilitation.

In the United States and Europe, dedicated energy crop yields of about 5 t C/ha/yr currently are achievable from good cropland, and yields of around 9 t C/ha/yr are believed possible by the year 2030 (Wright and Hughes, 1993). In Brazil, SRWC yields in the range of 20 to 30 t C/ha/yr have been reported from trial plots (Betters *et al*., 1992), representing a doubling of average yields from 30 years ago. Generally, however, average tropical crop yields are lower, ranging from 6 t C/ha/yr currently to perhaps 12 t C/ha/yr in the future (Table 23-4; Sampson *et al*., 1993).

Given the assumptions in Table 23-4, the range of estimated C emission reductions from energy crops in the tropics is 160 to 513 Mt C/yr. In the temperate regions, C emission reduction could potentially range from 85 to 493 Mt C/yr (Sampson *et al*., 1993). In addition, agroforestry systems, where trees are grown in intensively managed combinations with food or feed crops, have potential emission reductions of 10 to 55 Mt C/yr in temperate and 46 to 205 Mt C/yr in tropical regions.

#### *23.2.4.2. Biodiesel and Bioethanol*

Recently, the use of vegetable oil crops for the production of biodiesel has attracted considerable attention in the United States and the EU (Scurlock *et al*., 1993). Biodiesel can be burned directly in modified diesel engines or can be used in conventional diesel engines after conversion into methyl or ethyl esters (Vellguth, 1983; Schwab *et al*., 1987; Sims, 1990). However, it currently costs considerably more to produce than petroleum diesel, so it is not likely to see expanded usage unless technological breakthroughs, ecological considerations, or government subsidies alter the economic situation

***Table 23-4:*** *Potential reduction of C emissions when replacing fossil energy carriers by biofuels of agricultural origin under 2 x $CO_2$ conditions. Ranges indicate low and high estimates (modified from Sampson et al., 1993).*

| Agric. Biofuel Option | Land Area (Mha)[a] | Net C Yield (t C/ha/yr)[b] | Net C Amount (Mt/yr) | Energy Use (%)[c] | Energy Substitution Factor[d] | C Emissions Reduction (Mt/yr) |
|---|---|---|---|---|---|---|
| Dedicated Energy Crops | | | | | | |
| Temperate[e] | 26–73 | 5–9 | 130–657 | 100 | 0.65–0.75 | 85–493 |
| Tropical[f] | 41–57 | 6–12 | 246–684 | 100 | 0.65–0.75 | 160–513 |
| Temperate Shelterbelts[g] | 13–26 | 2–4 | 26–104 | 75 | 0.50–0.70 | 10–55 |
| Tropical Agroforestry[h] | 41–65 | 3–6 | 123–390 | 75 | 0.50–0.70 | 46–205 |
| **Total** | | | **525–1835** | | | **301–1266** |

[a]Assuming about 10–15% of world cropland to be available for biofuels. The 10% estimate agrees with Hall *et al*. (1993).
[b]Based on information in Flaig and Mohr (1994), Graham *et al*. (1993), Hall *et al*. (1993), Sampson *et al*. (1993), and others.
[c]Assumed percentage for energy utilization.
[d]Assumed substitution factors for fossil fuel according to Sampson *et al*. (1993).
[e-h]Assumed percentages of total cropland.
[e]8–11% temperate.
[f]5–7% tropical.
[g]2–4% temperate.
[h]5–8% tropical.

(Kleinhanss, 1993; Sampson *et al*., 1993; Scurlock *et al*., 1993; Flaig and Mohr, 1994).

Generally, crops from which only the oil, starch, or sugar are used are of limited value in reducing $CO_2$ emissions, due to the low net energy produced and the relatively high fossil fuel inputs required (Marland and Turhollow, 1991; Flaig and Mohr, 1994). Providing one energy equivalent as rapeseed oil requires about 0.5–0.6 equivalents as fossil fuel, and if bioethanol is produced from grains or root crops, the net energy gain may be as low as 13–20% (Leible and Wintzer, 1993; Graef *et al*., 1994). However, when using sugarcane, the cane waste provides most of the production energy, resulting in a ratio of ethanol energy to the input of fossil fuel of about 5.2 (Goldemberg *et al*., 1993). Similarly, palm oil obtained in mills driven by residue fuel at yields of 4–5 t/ha can compete with diesel costs at an energy output versus fossil energy input ratio of 9:5 (Wood and Conley, 1991). The burning of whole-plant biomass as an alternative to fossil fuel results in the most significant $CO_2$ mitigation (Leible and Wintzer, 1993; Reinhardt, 1993; Kaltschmitt and Becher, 1994), although the actual net effect depends on the plant yield and composition and on the intensity of the cropping system.

#### 23.2.4.3. Overall Fossil Fuel Offsets

Estimating biomass energy production potentials requires assumptions not only about available land, productivity, plant species, and percent of the crop to be used but also about collection and transport, conversion efficiencies, and fuel substitution factors. Table 23-4 was developed as an estimate of the primary energy that could be substituted over the next few decades as a result of agricultural biomass production. Assumptions were made about the relative conversion efficiency of individual fuels and regarding which fossil fuel was to be substituted for by the biomass (Sampson *et al*., 1993). Overall, agricultural biofuels (energy crops, agroforestry, and crop residues) have the potential to substitute for 0.40 to 1.50 Gt fossil fuel C per year.

### 23.2.5. Summary of $CO_2$ Mitigation in Agriculture

Potential C mitigation options in agriculture are significant in relation to anthropogenic emission rates (Table 23-5). The agricultural sector can reduce $CO_2$ increases in the atmosphere by reducing agriculturally related emissions, sequestering C in

***Table 23-5:*** *Summary of $CO_2$ mitigation potential for agriculture, expressed as decreases in net C emission rates or as net C storage rates, calculated on an annual basis or accumulated for a 50-year period.*

| Category | Annual (Gt C) | Cumulative (Gt C) |
|---|---|---|
| **Reducing C Emissions** | | |
| – Reduction in fossil energy use by agriculture in industrialized countries[a] (assuming 10–50% reduction in current use) | 0.01–0.05 | 0.5–2.5 |
| **Increasing C Sinks** | | |
| – Increasing soil C through better management of existing agricultural soils (globally)[b] | 0.4–0.6 | 22–29 |
| – Increasing soil C through permanent set-aside of surplus agricultural land in temperate regions | | |
| 1) Upland soils[c] | 0.015–0.03 | 0.75–1.5 |
| 2) Wetland restoration[d] | 0.006–0.012 | 0.3–0.6 |
| – Restoration of soil C on degraded lands[e] (assuming restoration of 10–50% of global total) | 0.024–0.24 | 1.2–12 |
| **Fossil C Offsets** | | |
| – Biofuel production from dedicated crops[f] | 0.3–1.3 | 15–65 |
| – Biofuel production from crop residues[g] | | |
| **Total Potential $CO_2$ Mitigation** | **0.86–2.44** | **45–122** |

[a] Based on current use of 3–4.5% of the total fossil C emission (2.8 Gt C/yr; OECD, 1991) by industrialized countries and an arbitrary reduction range of 10–50%.

[b] Assuming a recovery of one-half to two-thirds of the estimated historic loss (44 Gt) of C from currently cultivated soils (excluding wetland soils) over a 50-year period.

[c] Based on an estimated C sequestration of 1.5–3 Gt over a 100-year period, from a 15% set-aside of cultivated soils (~640 Mha), in industrialized countries with current or potential production surpluses; annual and cumulative rates given as 1 and 50% of that total, respectively.

[d] Based on restoration of 10–20% of former wetland area (8 Mha) now under cultivation in temperate regions.

[e] Assuming potential C sequestration of 1–2 kg C/$m^2$ over a 50-year period, on an arbitrary 10–50% of moderately to highly degraded land ($1.2 \times 10^9$ ha globally; Oldeman *et al*., 1990).

[f] Values from Table 23-4.

[g] Based on 25% recovery of crop residues and assumptions on energy conversion and substitution.

soils, and producing biofuel to replace fossil fuels. However, most of the options dealing with land use and soil C sequestration are limited in duration, in that vegetation and soils (under a given set of environmental and management conditions) have a finite capacity to sequester C. Calculations of C sink increases are based on estimates of the difference between current C stocks and those possible under improved management, considering that most of the increase in C would occur within a 50- to 100-year time frame. An exception is C accumulation in wetlands, where C increases can be sustained for much longer periods. Reductions in fossil C consumption by agriculture and the production of biofuels are mitigation options that can, in principle, be maintained indefinitely.

There is a high degree of uncertainty in our estimates concerning both flux rates and C storage capacity, as well as in the level at which various mitigation options could be implemented. Since the latter is largely a function of policy, several of the mitigation potentials have been expressed in terms of arbitrary ranges (10–50%) in implementation, representing possible lower and upper limits for developing policy scenarios.

Large amounts of C could be sequestered in soils (23–44 Gt over a 50-year period) through improved management of agricultural land, permanent set-asides, and restoration of degraded lands. Increasing soil C levels has additional benefits in terms of improving the productivity and sustainability of agricultural production systems. There are potential costs associated with promoting C storage, including fossil fuel requirements (e.g., fertilizers), lost production (e.g., set-aside programs), and additional labor and financial requirements (e.g., land restoration), which may constrain the potential for increasing C storage. Direct fossil fuel use by agriculture is a relatively minor portion of society's total consumption; therefore, even high reductions in use within agriculture have a modest mitigation potential.

The effects of potential climate change and $CO_2$ enrichment on mitigation have not been explicitly dealt with in the estimates of mitigation potentials. We recognize the importance of these factors, but the present level of knowledge is insufficient to provide quantitative estimates that would incorporate the complex interactions among $CO_2$, climate change, and land use and management. Analysis of the effects of these factors on C balance and trace-gas emissions requires an integrated model and data analysis at the global scale.

## 23.3. Methane and Nitrous Oxide

Successful development and implementation of mitigation strategies for agricultural sources of $CH_4$ and $N_2O$ require an understanding of the effects of land-use change and agricultural practices on fluxes of these gases and on controlling mechanisms. Current knowledge (Batjes and Bridges, 1992; Beese, 1994; Granli and Bockman, 1994; Kroeze, 1994; Mosier *et al.*, in press; Smith *et al.*, 1994) falls short of these criteria but is sufficient to identify key systems/practices and geographic areas to target, as well as likely mitigation technologies. Proposed mitigation options need to be evaluated within the context of farm production systems in order to ensure that interactions and/or feedbacks are accounted for. There is a potential for tradeoffs between $CH_4$ and $N_2O$ that are only partly assessed in this document.

Sections 23.3.1.1 and 23.3.1.2 provide a new assessment of agricultural sources and sinks of $CH_4$ and sources of $N_2O$. Appraisal of mitigation potential in agricultural systems is based on these estimates and the management technology available. The sources of $CH_4$ considered important include ruminant animals, rice production, animal waste, and biomass burning, while agricultural land use impacts the aerobic soil sink for atmospheric $CH_4$. The direct and indirect production of $N_2O$ in fertilized systems are considered to be the major agricultural source of $N_2O$. Other sources—such as biomass burning, enhanced production of $N_2O$ after burning, and increased $N_2O$ production after forest clearing for agriculture—are smaller and are given less rigorous attention. Animal feeding operations are considered minor $N_2O$ sources that are not yet adequately characterized.

### *23.3.1. Agricultural Sources and Sinks of $CH_4$ and $N_2O$*

#### *23.3.1.1. Methane*

Biological generation of $CH_4$ in anaerobic environments, including enteric fermentation in ruminants (Johnson *et al.*, 1993), flooded rice fields, and anaerobic animal waste processing, is the principal source of $CH_4$ from agriculture. Biomass burning associated with agriculture also contributes to the global $CH_4$ budget (Delmas, 1993). The primary sink for $CH_4$ is oxidation with hydroxyl radicals in the troposphere (Crutzen, 1981); in addition, an aerobic soil sink of 10–20% of $CH_4$ emissions is now evident (Reeburgh *et al.*, 1993).

##### *23.3.1.1.1. Ruminant animals*

Methane emissions from domestic ruminants are estimated to be about 80 Mt/yr, with a range of 65–100 Mt/yr (IPCC, 1992; Hogan, 1993). Cattle and buffalo account for about 80% of the global annual $CH_4$ emissions from domestic livestock (Hogan, 1993). Non-ruminant livestock make a relatively small contribution (Crutzen *et al.*, 1986; Gibbs and Leng, 1993; Johnson *et al.*, 1993).

Methane emissions associated with enteric fermentation in ruminants range from 3–8% of gross feed energy intake (Gaedeken *et al.*, 1990; Leng, 1991; U.S. EPA, 1992; Gibbs and Leng, 1993; Johnson *et al.*, 1993), producing 25–37 l of $CH_4$ per kg dry matter intake (Shibata, 1994). However, for the vast majority of the world's domestic ruminants consuming a wide range of diets under common production circumstances, $CH_4$ emissions fall near 6% of diet gross energy (range of 5.5 to 6.5%) (Johnson *et al.*, 1993). Restricting the amount of high-quality diet to one-half or less of voluntary consumption, as is frequently done experimentally, can double these percentage losses; however, this seldom occurs in practice. The only production group found markedly different from 6% is the approximately 25 million head of cattle fed very high concentrate diets for about 120 d prior to slaughter (primarily U.S. feedlots), where the emissions average about 3.5% of diet gross energy.

The relatively constant $CH_4$ emissions as percent of diet for most of the world's livestock has important implications for mitigation strategy. Strategies targeting increased available energy per unit of feedstuff will decrease intake required per unit of product (meat, milk, etc.). Improved feeding and animal management is the only strategy, to date, that has consistently reduced methane emissions. Almost all improved livestock management practices reduce methane per unit of product, and while increased diet energy availability is usually central to the improved practice, a whole array of interrelated inputs are required (protein, minerals, vitamins, improved genetics, reproductive efficiency, animal health, predator control, etc.). The trend worldwide is toward these improved technologies, limited mostly by the market price of animal products.

Examples of methane mitigations that have been analyzed include:

- A variety of feed additives (e.g., antibiotics, ionophore antibiotics and steroids) increase growth rate and increase the feed efficiency of beef cattle, resulting in 5–15% less methane per unit of product (U.S. EPA, 1992).
- Somatotropin (bST) to increase milk production, recently approved in the United States and in several other countries, is projected to reduce methane by 9% if adopted for all dairy cows in the United States (Johnson *et al*., 1991).
- Treatment of cereal straws with ammonia is a much-researched method and increases digestibility and intake, but increases in methane per kg of straw have been found (Birkelo *et al*., 1986).

Estimated potentials for adoption of these practices and associated $CH_4$ reductions are shown in Table 23-6. The greatest opportunity for reducing $CH_4$ emissions from ruminants is through feed supplementation of cattle and buffalo in Africa, Asia, and Latin America (Leng, 1991; Lin *et al*., 1994). Supplementation of the diets of native cattle/buffalo in India has been shown to decrease $CH_4$ emissions per liter of milk produced by a factor of three and per ton of live weight gain by a factor of six (Leng, 1991).

Opportunities for reducing $CH_4$ emissions from intensively managed cattle are somewhat limited because the $CH_4$ production per unit of cattle feed is small with a high-quality diet. For dairy cattle, estimated reductions are 10% by genetic improvement, 10% through use of bovine growth hormone (if permitted), and 4% through improved feed formulation (CAST, 1992; U.S. EPA, 1992). For beef cattle, pharmaceuticals being developed that promote protein gain at the expense of fat could reduce $CH_4$ emissions by as much as 20% (CAST, 1992). Longer-term opportunities include the production of twins to reduce the need for breeding animals and biotechnological approaches to modify rumen fermentation (CAST, 1992; U.S. EPA, 1992). Overall, the combination of these mitigation options could decrease $CH_4$ emissions from ruminant animals by approximately 30% (Table 23-6). However, increasing food quality may result in increased $N_2O$ emissions and increased energy use, thereby decreasing the effect of the estimated decreases in $CH_4$ emissions (Ward *et al*., 1993).

#### *23.3.1.1.2. Animal waste*

Significant emissions of $CH_4$ also occur from animal waste, varying with waste type and management practice (Safley *et al*., 1992; U.S. EPA, 1993). In general, manures from animals having a high-quality diet have higher potential to generate $CH_4$ than manures from animals having a low-quality diet. Actual $CH_4$ emission values depend on the amount of manure produced, its potential to generate $CH_4$, manure handling practices, and climate.

Global $CH_4$ emissions from livestock manure were an estimated 20–30 Mt/yr (Safley *et al*., 1992). More recently, the U.S. Environmental Protection Agency estimated this source to contribute 10–18 Mt/yr (Gibbs and Woodbury, 1993); a value of 14 Mt $CH_4$/yr is used here. Manure management systems that store manure under anaerobic conditions contribute about 60% of $CH_4$ of this source because $CH_4$ is produced during anaerobic decomposition of organic materials in manure (U.S. EPA, 1994). Methane lost from anaerobic digestion constitutes a wasted energy source that can be recovered by using manure management and treatment practices adapted for $CH_4$ collection (Hogan, 1993). With current technology, $CH_4$ emissions can be reduced by 25 to 80%. Hogan (1993) identified $CH_4$ control options that include (Table 23-6):

- **Covered Lagoons**—This option is associated with large-scale, intensive farm operations that are common in North America, Europe, and regions of Asia and Australia. Covered animal-waste lagoons have the potential to recover completely the $CH_4$ produced from anaerobic waste fermentation. A reasonable estimate is that 40% of the $CH_4$ from these regions could be mitigated using this approach.
- **Small-Scale Digesters**—These digesters are designed to utilize anaerobic decomposition of organic materials for $CH_4$ recovery, typically in small-scale operations. Wang *et al*. (1994) noted that about 10 million such biogas digesters are in use in China. These digesters also seem applicable to parts of Africa and South America. Here we assume an efficiency of 70%.
- **Large-Scale Digesters**—Larger, more technically advanced digesters can be integrated with management practices at large livestock operations.

***Table 23-6:*** *Estimated effect of management practices on $CH_4$ emissions from ruminant livestock, livestock manures, and flooded rice.*

| Mitigation Practice | | Estimated Decrease due to Practice (Mt $CH_4$/yr) |
|---|---|---|
| **Ruminant Livestock** | | |
| – Improving diet quality and nutrient balance | | 25 (10–35)[a] |
| – Increasing feed digestibility | | 2 (1–3) |
| – Production-enhancing agents | | 2 (1–6) |
| – Improved animal genetics | | – |
| – Improved reproduction efficiency | | – |
| | **Total** | **29 (12–44)** |
| **Livestock Manures** | | |
| – Covered lagoons | | 3.4 (2–6.8) |
| – Small digesters | | 1.7 (0.6–1.9) |
| – Large digesters | | – |
| | **Total** | **5.1 (2.6–8.7)** |
| **Flooded Rice** | | |
| – Irrigation management[e] | | 5[b] (3.3–9.9) |
| – Nutrient management | | 10[c] (2.5–15) |
| – New cultivars and other cultural practices | | 5[d] (2.5–10) |
| | **Total** | **20 (8–35)** |

[a] Range of estimates.

[b] About 50% of total rice area and 60% of total rice grain produced is under irrigation (Neue, 1992). Methane production is higher in continually flooded systems and with greater biomass production (Sass *et al*., 1991). Sass *et al*. (1992) showed that draining the field at specific times decreased $CH_4$ production by 88% without decreasing rice yield. The effect will probably not be better than 50% when used in the field. The ability to control flooding and drainage will be available to no more than 30% of the total production.

[c] Lindau *et al*. (1993) showed that $CH_4$ emissions can be decreased by adding sodium sulfate (by 28–35%) or coated calcium carbide (by 36%) with urea compared to urea alone, and by using ammonium sulfate (20%) in place of urea. With all these approaches, $CH_4$ production could be decreased by about 20%. We assume that this potential can be utilized for all rice production.

[d] Further mitigation potentials are likely to exist in optimizing rice cultivars and other management practices (Neue, 1992; Sass *et al*., 1992; Lin *et al*., 1994; Sass, 1994). In controlled experiments, a decrease of about 20% was demonstrated for some of these approaches. An estimated 10% reduction can be achieved from global rice production. Composting rice straw before field application also decreases $CH_4$ from rice fields (Yagi and Minami, 1990).

[e] Drying the soil surface during the cropping season may increase $N_2O$ emissions. The net effect of both gases needs to be evaluated before the practice can be recommended.

#### *23.3.1.1.3. Rice production*

The major pathways of $CH_4$ production in flooded soil are the reduction of $CO_2$ with $H_2$, fatty acids, or alcohols as hydrogen donor and the transmethylation of acetic acid or methyl alcohol by methane-producing bacteria (Conrad, 1989). In flooded rice fields, the kinetics of microbial reduction processes are strongly affected by the composition and texture of soil and its content of inorganic electron acceptors (Neue, 1992). The period between flooding and the onset of methanogenesis can be different for various soils (Sass *et al*., 1992). As methane production occurs only under highly reduced conditions, on the order of -200 mv redox potential, intermittent flooding or mid-season drainage decreases $CH_4$ emissions (Sass *et al*., 1992; Yagi and Minami, 1993). The amount of $CH_4$ produced in a system is highly dependent upon the quantity of organic carbon available, either from added rice straw (Schutz *et al*., 1989; Yagi and Minami, 1990; Sass *et al*., 1991; Neue *et al*., 1994; Nouchi *et al*., 1994b) or green manures (Lauren and Duxbury, 1993; Lauren *et al*., 1994). Rice plants influence $CH_4$ emissions by providing substrate for root exudation and decay (Schutz *et al*., 1989; Sass *et al*., 1991).

There are three processes of $CH_4$ release into the atmosphere from rice paddies. Methane loss as bubbles is generally a mechanism during the early stages of plant growth and during weeding operations. Diffusion loss of $CH_4$ across the water surface is another but is a relatively slow process. The third process, transport through rice plant aerenchyma and release to the atmosphere through the shoot nodes, which are not subject to stomatal control, is generally the most important emission mechanism (Cicerone *et al*., 1983; Seiler *et al*., 1984; Nouchi *et al*., 1990, 1994a, 1994b). During the course of the rice-growing season, a large portion of the $CH_4$ that is produced in the flooded soil is oxidized before it escapes to the atmosphere (Schutz *et al*., 1989; Sass *et al*., 1992).

*Revised emission estimates:* Minami (1993) proposed the methodology for estimating global $CH_4$ emission from rice fields and presented a $CH_4$ emission range of 10 to 113 Mt $CH_4$/yr from world rice paddy fields. A review of $CH_4$ studies in China, India, Japan, Thailand, the Philippines, and the United States (Sass, 1994) tightened the range of projected $CH_4$ emissions from rice fields. In his calculations, Sass combined the data for total area of rice paddies with the flux estimates published in various chapters of Minami *et al*. (1994) to produce Table 23-7. For China and India, the annual $CH_4$ flux estimates were specified by Wang *et al*. (1994) and Parashar *et al*. (1994), respectively. In the other cases, the figures are based on the minimum and maximum reported emission averages. The rice areas in the countries shown in Sass' estimate represent 63% of the total world rice paddy area and result in a total annual $CH_4$ emission of 16 to 34 Mt. Extrapolating these data to the world, Sass estimates total $CH_4$ emissions from rice fields to range between 25.4 and 54 Mt/yr, with 50 Mt/yr as a best-guess global emission value. This value is near the IPCC (1992) best estimate of 60 Mt $CH_4$/yr, but the range indicates that the actual rate may be lower. Wang *et al*. (1994) estimated the annual $CH_4$ emissions from rice in China to range between 13 and 17 Mt/yr. Lin *et al*. (1994) estimated a slightly lower value from rice in China of <12 Mt $CH_4$/yr.

***Table 23-7:*** *Estimates of methane emissions from rice fields.*

| Country | Total Area of Rice Paddies (Mha) | Total Rice Grain Yield (Mt/yr) | $CH_4$ Emission (Mt/yr) |
|---|---|---|---|
| China | 32.2 | 174.7 | 13–17[a] |
| India | 42.2 | 92.4 | 2.4–6[b] |
| Japan | 2.3 | 13.4 | 0.02–1.04[c] |
| Thailand | 9.6 | 19.2 | 0.5–8.8[d] |
| Philippines | 3.5 | 8.9 | 0.3–0.7[e] |
| USA | 1.0 | 6.4 | 0.04–0.5[f] |
| Other | 54.6 | 158.5 | 9.2–20 |
| **World Total** | **147.5** | **473.5** | **25.4–54**[g] |

[a] Wang *et al*. (1994). $CH_4$ values based on an area-weighted summation for specific regions of the country.
[b] Parashar *et al*. (1994). $CH_4$ calculated as in (a).
[c] Yagi *et al*. (1994). $CH_4$ values based on measured minimum and maximum emission rates from the country.
[d] Yagi *et al*. (1994). $CH_4$ calculations as in (c).
[e] Neue *et al*. (1994). $CH_4$ calculations as in (c).
[f] Sass and Fischer (1994). $CH_4$ calculations as in (c).
[g] World total emission rate obtained by an area scaling of the total of the emission rates measured (Sass, 1994).

*Estimates of $CH_4$ emission reduction:* Applying the following major management options to global rice production could decrease $CH_4$ production in rice: (1) water management, (2) nutrient management, (3) cultural practices, and (4) new rice cultivars (Table 23-6).

*23.3.1.1.4. Biomass burning*

The burning of biomass results in the emission of $CH_4$ because of incomplete combustion. Emission factors for $CH_4$ (i.e., the fraction relative to emitted $CO_2$) vary greatly (0.1% to 2.5%) depending on whether the fire is hot, flaming, or smoldering combustion (Levine *et al*., 1993). In response to declining agricultural yields and increasing population pressures, farmers in many regions convert forests to cropping land, and many of their techniques involve burning. For example, shifting cultivation requires that forests be cut, and logging debris and unwanted vegetation burned; the land is then farmed for several years, then left fallow to rejuvenate. Savanna and rangeland biomass is often burned to improve livestock forage. Agricultural residues are also burned in the field to return nutrients to the soil or reduce shrubs on rotational fallow lands. Such agriculture-related burning may account for 50% of the biomass burned annually. In addition, about 50% of the worldwide crop residues are burned in small-scale cooking and heating stoves (Hao *et al*., 1988).

Estimates indicate that 8,700 Mt dry matter/yr (Andreae, 1991) of biomass and 1 to 5% of the world's land (U.S. EPA, 1990b) is burned. Of these estimated $CH_4$ emissions, those from tropical forest clearing for agriculture, savanna burning, and agricultural crop residue burning are portions of total biomass burning that can be attributed to agricultural practices. Those sources total about 22 Mt $CH_4$/yr. Burning of crop lands, grasslands, and forests may be reduced through sustained land management programs and the promotion of different land-use practices, including the following:

- Increasing the productivity of existing agricultural lands
- Lengthening the rotation times and improving the productivity of shifting agriculture
- Increasing grassland management
- Incorporating crop residues into soil
- Increasing the use of crop residues as household fuel (remembering that a balance between fuel use and maintenance of soil fertility must be maintained)
- Replacing annual or seasonal crops with trees.

Using various combinations of these practices, an estimated 50% of the $CH_4$ emitted annually from burning of agricultural wastes (~2.7 Mt/yr) and 20% of other agricultural burning (~3.3 Mt/yr) could be eliminated, providing a total potential mitigation of ~6 Mt $CH_4$/yr.

*23.3.1.1.5. Methane oxidation in soil*

Land-use changes and other human-induced alterations of C and N cycles during the past centuries appear to have decreased

$CH_4$ oxidation in aerobic soils (Ojima *et al*., 1993), increased N deposition on temperate forest soils (Steudler *et al*., 1989), and decreased $CH_4$ uptake by 30 to 60%. Methane oxidation was decreased by about 50% by tilling a semi-arid grassland even when no N fertilizer was ever applied (Mosier *et al*., 1991). The decrease in $CH_4$ oxidation in soils when forests or grasslands are converted to agricultural use has been observed in tropical (Keller *et al*., 1990, 1993) and temperate (Steudler *et al*., 1989; Mosier *et al*., 1991; Dorr *et al*., 1993; Dobbie and Smith, 1994) environments. The decrease seems to be greater as the intensity of the agricultural practices increases. Ojima *et al*. (1993) estimated that land-use changes during the past 200 years have decreased the global temperate soil sink for $CH_4$ by 20–30%.

A portion of this effect can be attributed to inhibition of $CH_4$ uptake by inorganic N (Steudler *et al*., 1989; Keller *et al*., 1990; Mosier *et al*., 1991; Nesbit and Breitenbeck, 1992; Hütsch *et al*., 1993). Additions of inorganic N have been shown to reduce $CH_4$ uptake in many, but not all, cases (Steudler *et al*., 1989; Mosier *et al*., 1991; Adamsen and King, 1993; Cochran *et al*., 1995).

Reeburgh *et al*. (1993) estimated the global aerobic soil sink to be about 40 Mt $CH_4$/yr. From a review of available $CH_4$ uptake data, Minami *et al*. (1993) constrain total terrestrial $CH_4$ consumption between 7 and 78 Mt/yr. Insufficient information is currently available to recommend agricultural practices to increase the oxidation of $CH_4$ in cultivated soils.

### 23.3.1.2. Nitrous Oxide

Nitrous oxide is produced primarily by microbial processes in the soil (Bouwman, 1990). Anthropogenic emission of $N_2O$ occurs as a result of land conversion to agriculture and is likely to be most intensive in agricultural systems that have high N input. Because soil production is the major agricultural source of $N_2O$, this topic is emphasized (Granli and Bockman, 1994). Agricultural $N_2O$ emissions are thought to arise from fertilization of soils with mineral N and animal manures (this N is partly recycled mineral N and relocated soil mineral N), N derived from biological N fixation (legume crops and free-living N-fixing microbes), and from enhanced soil N mineralization (Duxbury and Mosier, 1993). Information is available only to assess the first three sources (Mosier and Bouwman, 1993; Isermann, 1994a). Nitrous oxide also is directly evolved during biomass burning, and produced in soil after burning, and enhanced emissions arise during conversion of tropical forest to agriculture (Batjes and Bridges, 1992).

A variety of factors control rates of the two microbial processes (nitrification and denitrification) that produce $N_2O$ and $N_2O$ yield. Important variables are soil water content, temperature, nitrate or ammonium concentrations, available organic carbon for denitrification, and pH. Because interactions among the physical, chemical, and biological variables are complex, $N_2O$ fluxes from agricultural systems are highly variable in both time and space (Duxbury and McConnaughey, 1986; Smith, 1990; Beese, 1994; Clayton *et al*., 1994; Kroeze, 1994; McTaggart *et al*., 1994). Consequently, prediction of $N_2O$ emissions associated with a unit of N applied to a specific field or fixed by legumes (Mosier, 1993) is not yet reliable. Such predictive capabilities are needed because $N_2O$ emissions derived from agriculture are >75% of the anthropogenic sources (Isermann, 1994a).

#### 23.3.1.2.1. Revised $N_2O$ emission estimates from agricultural soils

Estimates of N inputs to agricultural soils and associated $N_2O$ production are presented in Tables 23-8 and 23-9, respectively. The data are grouped into seven regions of the world. These estimates consider both $N_2O$ directly emitted from agricultural fields and the indirect emissions that occur during parts of the year other than the cropping season, after the N leaves the field. The N fixation contribution does not include $N_2O$ produced in legume pastures. Australia and New Zealand, for

***Table 23-8:*** *Estimated nitrogen applied annually to agricultural lands as synthetic fertilizers and animal wastes, and land area cropped with pulses and soybeans.*

| | **Synthetic N Consumed** | **Manure N Produced** | **Manure N Used as Fertilizer** | | **Harvested Area of Pulses + Soybeans** |
|---|---|---|---|---|---|
| **Region** | Mt | Mt | % of Total Manure N | Mt | ha x $10^6$ |
| Africa | 2.1 | 20.9 | 50 | 10.5 | 12.8 |
| North and Central America | 13.1 | 7.8 | 70 | 5.5 | 28.1 |
| South America | 1.7 | 21.9 | 50 | 11.0 | 23.5 |
| Asia | 37.3 | 37.4 | 70 | 26.2 | 48.5 |
| Europe | 13.6 | 12.3 | 90 | 11.1 | 4.0 |
| Oceania | 0.9 | 0.5 | 30 | 1.5 | 1.4 |
| Former Soviet Union | 8.7 | 10.1 | 90 | 9.1 | 6.6 |
| **Total** | **77.4** | **115.3** | | **74.9** | **124.9** |

***Table 23-9:*** *Estimates of direct and indirect emissions of $N_2O$ from application of fertilizer N (synthetic or animal waste) to agricultural soils and from soils growing biological N-fixing crops (Mt $N_2O$-N/yr).*

| | **Estimated $N_2O$ from** | | | | |
|---|---|---|---|---|---|
| **Region** | Mineral N | Animal Waste | N-Fixation | **Total** | **Range** |
| Africa | 0.04 | 0.21 | 0.05 | 0.30 | 0.15–0.45 |
| North and Cental America | 0.26 | 0.11 | 0.11 | 0.48 | 0.24–0.72 |
| South America | 0.03 | 0.22 | 0.09 | 0.34 | 0.17–0.51 |
| Asia | 0.75 | 0.52 | 0.19 | 1.46 | 0.73–2.19 |
| Europe | 0.27 | 0.22 | 0.02 | 0.51 | 0.26–0.77 |
| Oceania | 0.01 | 0.03 | 0.01 | 0.05 | 0.03–0.08 |
| FSU | 0.17 | 0.18 | 0.03 | 0.30 | 0.19–0.57 |
| **Total** | **1.53** | **1.49** | **0.50** | **3.50** | **1.8–5.3** |

example, contain large areas of pasture land that include legumes as part of the pastoral system. Little data are available for other parts of the globe.

The estimate of synthetic fertilizer N inputs to agricultural soils and the land area of harvested pulses and soybeans are based on country data published by FAO (1990a, 1990b). These values are subject to uncertainty, as are the highly uncertain animal manure N values adapted from Safley (1992) and Bouwman *et al*. (in press). The quantities used as fertilizer for each region are shown in Table 23-8. These numbers are based on estimates of animal distribution and management systems for each region and are not very reliable because of the lack of information.

In addition to including multiple N input sources and $N_2O$ derived from N-fixation, a revised method for calculating the contribution of $N_2O$ from agriculture is used for the estimates in Table 23-9. Earlier estimates generally were based upon assessments derived from reviews of published $N_2O$ emissions data (Bouwman, 1990; Eichner, 1990). More recently, Bouwman *et al*. (in press) reviewed the literature again and presented another assessment of $N_2O$ emissions. They noted that loss of $N_2O$ from agricultural soils may be presented in three ways: (1) the total loss during the period covered by the measurements; (2) the difference between fertilized and control plot, which is referred to as "fertilizer-induced $N_2O$ loss"; and (3) the total loss calculated as a percentage of fertilizer N applied.

Bouwman *et al*. (in press) estimated the total emissions of $N_2O$ from a regression equation: total annual direct field $N_2O$-N loss = 1 + 0.0125 * N-application (kg N/ha). The value of 1 kg $N_2O$-N/ha represents the background $N_2O$-N evolved; the 0.0125 factor accounts for the contribution from fertilization. This estimate includes N sources from a variety of mineral and organic N fertilizers and was based on long-term data sets. The total flux represents $N_2O$ from all sources: native soil N, N from recent atmospheric deposition, past years' fertilization, N from crop residues, $N_2O$ from subsurface aquifers, and current N fertilization.

As explained in Mosier (1993), soil management and cropping systems and unpredictable rainfall inputs affect $N_2O$ emissions more than mineral N sources. As a result, for the purpose of estimating $N_2O$ production from fertilization, we do not use different multiplication factors for different fertilizer types. Limited data also indicate that organic N sources such as animal manures and sewage sludge induce larger $N_2O$ emissions per unit of N added to the soil than does mineral N (Bouwman, 1990, 1994a; Benckiser and Simarmata, 1994). Because of the lack of adequate parallel experiments that cover the range of possibilities of mineral and organic N applications, a single conversion coefficient is used for all sources.

Although these emission estimates are variable, the range is lower than suggested in the OECD/OCDE (1991) calculation methodology. Experience in conducting field flux experiments suggests that much narrower constraints can be placed on the $N_2O$ flux predictions. Bouwman (1994a) estimated that 1.25 ± 1.0% of the applied N was directly emitted, as $N_2O$ encompasses approximately 90% of the direct contributions of fertilization to $N_2O$ emissions.

The indirect contribution of fertilizer N to $N_2O$ emissions, apart from the fields where fertilizer is applied, also must be considered. Based upon the discussions of Duxbury *et al*. (1993), Mosier (1993), and Isermann (1994a), and the large amounts of $N_2O$ frequently found in subsurface aquifers (Bowden and Bormann, 1986; Minami and Ohsawa, 1990), an estimated additional 0.75% of N applications will eventually be evolved to the atmosphere as $N_2O$ resulting from N leaching, runoff, and nitrogen oxides ($NO_x$) and ammonia ($NH_3$) volatilization.

The direct and indirect $N_2O$-N emissions from application of mineral or organic N total approximately 2 ± 1% annually (Mosier *et al*., in press). This estimate is expected to encompass more than 90% of field situations. Nitrous oxide from biological N-fixation is calculated by multiplying the area of land used for growing pulses plus soybeans in each region by 4 kg N/ha (Duxbury *et al*., 1982; Galbally *et al*., 1991).

*23.3.1.2.2. Mitigation of $N_2O$ from agricultural soils*

A significant fraction of the $N_2O$ evolved from agricultural systems could be avoided if some combination of agricultural management practices listed in Table 23-10 were adopted worldwide. These practices are recommended mainly to improve synthetic fertilizer and manure N use efficiency. The underlying concept in limiting $N_2O$ emissions is that if fertilizer N (all N applied to improve crop growth) is utilized better by the crop, the amount of N needed to meet the growing demand for food will be less; therefore, less $N_2O$ will be produced and less N will leak from the system (Isermann, 1994b; Sauerbeck, 1994a, 1994b). Some of these practices, such as use of nitrification inhibitors, have been shown to have a direct effect on decreasing $N_2O$ emissions in field studies (Aulakh *et al*., 1984; Minami *et al*., 1990; Bronson *et al*., 1992). Ryden (1981) and McTaggart *et al*. (1994) have shown that timing of application of different types of synthetic fertilizer with seasonal water distribution can limit $N_2O$ production.

From Table 23-10 the amount of $N_2O$-N that is amenable to management is estimated. In these calculations, it is assumed that two-thirds of the $N_2O$ from N applied as manure or synthetic fertilizer is directly emitted from agricultural systems. The remaining one-third is emitted indirectly as a result of runoff, nitrate leaching, and transfer of N to other sections of the ecosystem through $NH_3$ and $NO_x$ emissions. Only the direct emissions are readily amenable to control by on-farm management, but management options that decrease the amount of external N needed to produce a crop also will decrease indirect $N_2O$ production. It was also assumed that $N_2O$ produced directly from biological N fixation cannot be managed. These estimates were based upon estimated fertilizer use and animal N production in FAO (1990b); thus, they represent estimates for that year and are not future projections.

By better matching N supply to crop demand and more closely integrating animal waste and crop-residue management with crop production, $N_2O$ emissions could be decreased by about 0.38 Mt $N_2O$-N. Further improvements in farm technology, such as use of controlled-release fertilizers, nitrification inhibitors, timing, and water management, should lead to improvements in N use efficiency and further limit $N_2O$ production by an estimated 0.30 Mt $N_2O$-N. A total potential reduction of global $N_2O$ emissions from agricultural soils is thus 0.7 (0.34 to 1.0) Mt $N_2O$-N/yr.

*23.3.1.2.3. Release due to biomass burning*

Bouwman (1993, 1994b) reviews the status of $N_2O$ formation during biomass burning and estimates a global emission of 0.1 to 0.3 Mt N/yr. This calculation is based on 0.7 ± 0.3% of the N content of the material burned being lost as $N_2O$ (Lobert *et al*., 1990; Hurst *et al*., 1994). The estimate includes only those emissions related to savanna burning and deforestation (Crutzen and Andreae, 1990; Hao *et al*., 1990). A mean value of 0.2 Mt N/yr to represent the $N_2O$ emitted directly from biomass burning related to agriculture is used here. The mitigation potential is difficult to assess because if crop residue is returned to the soil, part of the N mineralized will be converted to $N_2O$. A significant

***Table 23-10:*** *List of practices to improve efficiency of use of synthetic fertilizer and manure N in agriculture and expected reduction of $N_2O$ emissions assuming global application of mitigation practices (Mt N/yr).*

| **Practice Followed** | **Estimated Decrease in $N_2O$ Emissions** |
|---|---|
| **Match N Supply with Crop Demand** | 0.24[a] |
| – Use soil/plant testing to determine fertilizer N needs | |
| – Minimize fallow periods to limit mineral N accumulation | |
| – Optimize split application schemes | |
| – Match N application to reduced production goals in regions of crop overproduction | |
| **Tighten N Flow Cycles** | 0.14[b] |
| – Integrate animal and crop production systems in terms of manure reuse in plant production | |
| – Maintain plant residue N on the production site | |
| **Use Advanced Fertilization Techniques** | 0.15[c] |
| – Controlled-release fertilizers | |
| – Place fertilizers below the soil surface | |
| – Foliar application of fertilizers | |
| – Use nitrification inhibitors | |
| – Match fertilizer type to seasonal precipitation | |
| **Optimize Tillage, Irrigation, and Drainage** | 0.15[d] |
| **Total** | **0.68** |

[a] Assumed that fertilizer N use efficiency can be increased to save 20% of N applied in North America, Europe, and FSU (Doerge *et al*., 1991; CAST, 1992; Isermann, 1994a; Peoples *et al*., 1995).

[b] Tightening N cycles may decrease the need for 20% of the N that is used currently in North America, Europe, and FSU, thus saving 20% of fertilizer and reducing $N_2O$ from manure by the same amount where applicable (Buresh *et al*., 1993; Isermann, 1994a).

[c] Controlled-release fertilizers (Minami, 1994), nitrification inhibitors (Bronson *et al*., 1992; Keerthisinghe *et al*., 1993; McTaggart *et al*., 1994; Minami, 1994), and matching fertilizer type with seasonal precipitation can decrease $N_2O$ emissions 40–90%. We assume that 10% of all fertilizer-derived $N_2O$ production can be decreased by 50%.

[d] There is little published data to confirm this assumption (Granli and Bockman, 1994). A conservative assumption of a 5% decrease that can be achieved globally is used.

decrease in the amount of agricultural biomass that is burned could be achieved by composting the material before it is returned to the field. It is not known, however, how much $N_2O$ is released during composting. Burning also may make N and other nutrients more available to soil microorganisms and result in enhanced emissions of $N_2O$ from soil (Anderson *et al*., 1988; Anderson and Poth, 1989). Bouwman (1993) and Bouwman *et al*. (in press) calculate that from about 12 Mt N/yr remaining on the ground after burning (Crutzen and Andreae, 1990), 20% is volatilized as $NH_3$ and that 1%, 0.1 Mt/yr, of the remaining N is emitted to the atmosphere as $N_2O$. Because of this uncertainty, we expect a mitigation potential of about 10% of the $N_2O$-N associated with burning of biomass from agriculture.

##### *23.3.1.2.4. Conversion of tropical forest*

Conversion of tropical forests to pastures and arable land may contribute an important amount of $N_2O$ to the atmosphere (Keller *et al*., 1993; Bouwman *et al*., in press). While fluxes may increase by a factor of 5–8 in the first few years after forest clearing (Luizao *et al*., 1989; Keller *et al*., 1993), fluxes gradually decrease during the following 10–20 years (Garcia-Mendez *et al*., 1991; Keller *et al*., 1993). Bouwman (1994b) estimates that about 0.4 Mt of additional $N_2O$-N is emitted annually. We estimate that a 20% reduction from existing levels can be achieved.

### *23.3.2. Summary of Methane and Nitrous Oxide Emissions and Potential Decreases*

Mitigation options are available that could result in significant decreases in $CH_4$ and $N_2O$ emissions from agricultural systems. If implemented, they are likely to increase rather than decrease crop and animal productivity. Implementation has the potential to decrease $CH_4$ emissions from rice, ruminants, and animal waste by 30–40% (Table 23-11). The key to decreasing $N_2O$ emissions is improving the efficiency of plant utilization of fertilizer N. This could decrease $N_2O$ emissions from agriculture by almost 20%. Using animal waste to produce $CH_4$ for energy and digested manure for fertilizer may at some time be cost effective. Economic analyses of options proposed should show positive economic as well as environmental benefits.

## 23.4. Economic Feasibility of Mitigation Options

As shown in the preceding sections of this chapter, there are policies and tools that, if put into effect, can reduce net carbon emissions from agriculture and increase sequestration. Carbon emissions and capture in agriculture can be brought into balance, but agriculture always will be a net source of $N_2O$ and $CH_4$. Practices are described in this chapter that would reduce these emissions but not eliminate them. In the case of $N_2O$, these practices include changes in the timing and placement of fertilizer and use of nitrification inhibitors and fertilizer forms that slow the release of N. In the case of $CH_4$, these practices include shortening the time during which rice paddies are inundated, altering feeding and husbandry practices to diminish emissions per unit of animal product produced, and utilizing animal wastes for biogas production.

Implementation of these practices will require decisions at many different levels. For example, development of a biofuel

***Table 23-11:*** *Estimated potential impact of mitigation options on $CH_4$ and $N_2O$ emissions from agriculture.*

| **Source** | **Estimated Amount Emitted** (Mt/yr) | **Potential Decrease** (Mt/yr) | (%) |
|---|---|---|---|
| **$CH_4$** | | | |
| – Ruminant animals | 80 (65–100)[a] | 29 (12–45) | 36 |
| – Animal waste | 14 (10–18) | 3 (2–7) | 21 |
| – Rice paddies | 50 (20–60) | 20 (8–35) | 40 |
| – Biomass burning | 22 (11–33) | 6 (1.5–4.5) | 27 |
| **Total** | **166 (106–211)** | **58 (24–92)** | **35** |
| **$N_2O$-N** | | | |
| – Mineral fertilizer | 1.5 (0.5–2.5) | 0.3 (0.15–0.45) | 20 |
| – Animal wastes | 1.5 (0.5–2.5) | 0.3 (0.15–0.45) | 20 |
| – N-fixation | 0.5 (0.25–0.75) | – | – |
| – Biomass burning | 0.2 (0.1–0.3) | 0.02 (0.01–0.03) | 10 |
| – Soils after burning | 0.1 (0.05–0.2) | 0.01 (0.005–0.015) | 10 |
| – Forest conversion | 0.4 (0.1–1) | 0.08 (0.04–0.12) | 20 |
| **Total** | **4.2 (1.5–7.25)** | **0.71 (0.36–1.1)** | **17** |

[a]Range of estimates.

industry will require changes in infrastructure, institutions, and regional and national policies. At a different scale and level of decisionmaking, the rancher can help reduce $CH_4$ emissions by providing his/her animals with feed additives. Even the nomadic herder can do this by providing urea-containing blocks for grazing animals to lick. What are the incentives and disincentives for them to do so? Here we examine the feasibility for agricultural producers and the likelihood that they will adopt GHG reduction methods described in this chapter.

It seems reasonable that the world's farmers, ranchers, and pastoralists will not volunteer to implement practices proposed to mitigate greenhouse-forced climate change. This will happen only if the producer is convinced that profitability will improve if these practices are implemented. Incentives such as subsidies can be created to encourage their adoption, and penalties for nonadoption can be imposed.

Examples of mitigation efforts include the following:

- No-till agriculture that increases C storage in the soil is one example of a GHG-mitigating practice that meets the criteria for successful, unforced adaption. No-till practices are used increasingly in the United States in the production of corn, soybeans, and wheat. No-till accounted for about 10–14% of the total acreage of these crops in 1992, double what it had been 5 years earlier.
- Nutrient management practices that result in lower N application rates should reduce emission of $N_2O$. Soil testing, fertilizer placement, timing, class of fertilizer, and inhibitors are practices that supply nutrients in better accordance with plant physiologic demands. These practices are more likely to be economically feasible on crops that have high N demands (e.g., corn, cotton, and wheat). Because these practices already are being adopted to some extent, only minimal institutional programs may be needed to significantly increase their level of use.
- Reducing the number of days that rice fields are flooded will require large land and water resource investments to provide supplemental storage to change time of flooding. Economic feasibility will be dependent on many unique characteristics of the project site. Such cost estimates will need to address financing and repayment of any structural developments and how such developments change hydrology of water systems and impact other users, in addition to adoption costs incurred by individual producers. As there are no obvious on-farm benefits, and there are adoption costs for this practice, institutional intervention to provide economic incentives or a mandate requiring the practices will likely be needed.
- Increasing quality in livestock feeds is recommended as a means of reducing $CH_4$ emission per unit of animal product. In developed economies, where there is a high consumption of red meats and dairy products, most livestock animals already are fed higher quality roughage and concentrates. Producers have the knowledge and technical expertise to improve the quality of feed if it can be shown to increase their profits. In developing or underdeveloped countries, land needed for higher-quality roughage production often competes with the food needs of humans. Livestock rely on crop residue and other coarse roughage produced on land not suited for more intensive uses. To add concentrates to feed and to supplement pasture and range with improved crop species adds to production costs. Also, land resources may not be able to support improved species. Whether the benefits generated exceed these costs can best be determined through cost/benefit case studies involving local conditions. Local production relationships, input prices, and local markets will affect long-term economic feasibility.

In general, practices that recover investment cost and generate a profit in the short term are preferred over practices that require a long term to recover investment costs (Rahm and Huffman, 1984). Practices that have a high probability associated with expected profits are desired over practices that have less certainty about their returns. When human resource constraints or knowledge of the practice prevent adoption, public education programs can improve the knowledge and skills of the work force and managers to help advance adoption. Crop insurance or other programs to share the risk of failure due to natural disaster can aid the adoption of practices that increase productivity or expected returns.

## 23.5. Uncertainties and Future Research Needs

Uncertainties in our present assessment stem from two main sources. Both of these sources point to future research needed for improving our assessment of mitigation options. One source is the inherent unpredictability of future conditions that are controlled primarily by social, economic, and political forces. These conditions include such things as future trends in fossil fuel usage and the degree to which various mitigation strategies will be implemented—decisions that hinge on factors such as economic conditions, environmental awareness, and political will. The most effective way of dealing with these uncertainties is by developing analytical tools that can incorporate socioeconomic factors as potential scenarios in model gaming exercises. Currently, such tools for agricultural systems on a global basis do not exist.

The second source of uncertainty involves deficiencies in our scientific understanding of GHG processes, as well as inadequacies in the information base needed to apply the knowledge that we do have. Gaps in basic understanding can only be addressed through basic research; however, our current ability to assess GHG mitigation in agricultural systems is probably more constrained by a lack of baseline data, in an organized and usable format, than it is by insufficient scientific understanding. Thus, the research needs outlined below focus on the need for compiling and analyzing baseline information:

- **Carbon Sequestration in Agricultural Systems**—The assessment of potentials to increase C stocks in agricultural systems could be improved by using a structured, model-based analysis with global coverage. Suitable models for such a task currently exist and have been used for regional-level analyses (see Section 23.2.2.3). The elements that are lacking are: (1) spatial databases linking climate, soils, and land use and management, which are needed as model inputs; and (2) reliable experimental data to calibrate and/or verify model predictions. A compilation of agricultural land-use information to develop a classification and mapping of agroecological/management zones for the world is sorely needed. Existing long-term agricultural experiments can provide information to evaluate model predictions for different management systems, and soil and climate conditions. Efforts are underway to establish networks of long-term experiment sites and data (Paul *et al.*, 1995; Powlson *et al.*, 1995), but they are still at an early stage.
- **Bioenergy Production from Agricultural Lands**—Research needed to improve assessments of $CO_2$ mitigation potential through increased use of biofuels includes better technical knowledge of biofuel production and energy conversion efficiencies, as well as information on socioeconomic factors affecting the utilization of biofuels. These needs include: (1) improved information on the actual C feedstock value of forest, agroforestry, and agricultural management systems; (2) better data on energy inputs for the production of wooden goods and tree-derived chemicals and their substitutes; (3) better data on land availability, including cultural, social, and political factors that may preclude some lands from use for C offset projects; and (4) better data, including economic analyses, for the use and efficiency of biofuels, particularly where that usage is conducted outside markets.
- **$CH_4$ and $N_2O$ Emissions from Agricultural Systems**—Research needed to improve assessments of $CH_4$ and $N_2O$ mitigation options includes improved synthesis and coordination of existing information and additional field measurements:
  - Available field emission/consumption data need to be carefully assimilated so that comparisons of data sets can be made on a uniform basis. Those data sets that were collected over an insufficient period of time, used inadequate methodology, or are from nonrepresentative systems should be omitted.
  - Existing data need to be applied to validate and calibrate process-based models. Model estimates of gas fluxes should incorporate soil, cropping system, climate, and fertilizer management influences.
  - Field data on gas fluxes are still woefully inadequate. Research needs include year-round field flux measurements in a variety of soils, climates, and cropping systems to compare the impact of management on gas fluxes and to determine the tradeoffs between $CH_4$ and $N_2O$ flux when management options are exercised. Assessment of entire cropping sequences (e.g., rice-wheat-rice) are needed. The combined use of different flux measurement techniques is needed to evaluate systems over time and space.

Efforts to improve national, regional, and global estimates of gas fluxes are best accomplished through combined efforts. Unfortunately, other than for organizational exercises, little national or international funding has become available to conduct the needed research.

## References

**Adamsen**, A.P.S. and G.M. King, 1993: Methane consumption in temperate and subarctic forest soils: rates, vertical zonation, and responses to water and nitrogen. *Appl. Environ. Microbiol.*, **59**, 485-490.

**Amentano**, T.V. and E.S. Menges, 1986: Patterns of change in the carbon balance of organic soil-wetlands of the temperate zone. *Ecology*, **74**, 755-774.

**Andreae**, M.O., 1991: Biomass burning: its history, use, and distribution and its impact on environmental quality and global climate. In: *Global Biomass Burning: Atmospheric, Climate, and Biospheric Implications* [Levine, J.S. (ed.)]. MIT Press, Cambridge, MA, pp. 3-21.

**Aulakh**, M.S., D.A. Rennie, and E.A. Paul, 1984: Gaseous nitrogen losses from soils under zero-till as compared with conventional-till management systems. *J. Environ. Qual.*, **13**, 130-136.

**Balesdent**, J., A. Mariotti, and D. Boisgontier, 1990: Effect of tillage on soil organic carbon mineralization estimated from $^{13}C$ abundance in maize fields. *J. Soil Sci.*, **41**, 587-591.

**Batjes**, N.H. and S.M. Bridges (eds.), 1992: World inventory of soil emission potentials. Proc. Int. Workshop WAG, 24-27 August 1992, WISE-Report 2, Internat. Soil Reference and Information Centre, Wageningen, 122 pp.

**Beese**, F., 1994: Gasfoermige Stickstoffverbindungen. In: Studienprogramm Vol. 1 Landwirtschaft, Project D [Enquete Kommission "Schutz der Erdatmosphaere" des Deutschen Bundestages (ed.)]. Economica Verlag GmbH, Bonn, Germany, 88 pp.

**Benckiser**, G. and T. Simarmata, 1994: Environmental impact of fertilizing soils by using sewage and animal wastes. *Fertilizer Research*, **37**, 1-22.

**Betters**, D., L.L. Wright, and L. Couto, 1992: Short rotation woody crop plantations in Brazil and the United States. *Biomass and Bioenergy*, **1**, 305-316.

**Bockman**, O.C., O. Kaarstad, O.H. Lee, and I. Richards, 1991: Energieverbrauch in der Landwirtschaft. In: *Planzenernaehrung und Landwirtschaft*. Nask Hydro, Oslo, Norway, pp. 177-183.

**Bouwman**, A.F., 1990: Global distribution of the major soils and land cover types. In: *Soils and the Greenhouse Effect* [Bouwman, A.F. (ed.)]. John Wiley and Sons, Chichester, UK, pp 47-59.

**Bouwman**, A.F., 1993: The global source distribution of nitrous oxide. In: *Methane and Nitrous Oxide* [van Amstel, A.R. (ed.)]. RIVM Report No. 481507003, National Institute of Public Health and Environmental Protection, Bilthoven, Netherlands, pp. 261-272.

**Bouwman**, A.F., 1994a: *Method to Estimate Direct Nitrous Oxide Emissions from Agricultural Soils*. Report 773004004, National Institute of Public Health and Environmental Protection, Bilthoven, Netherlands, 28 pp.

**Bouwman**, A.F., 1994b: Estimated global source distribution of nitrous oxide. In: *$CH_4$ and $N_2O$: Global Emissions and Controls from Rice Fields and Other Agricultural and Industrial Sources* [Minami, K., A. Mosier, and R. Sass (eds.)]. NIAES Series 2, Yokendo Publishers, Tokyo, Japan, pp. 147-159.

**Bouwman**, A.F., K.W. van Der Hoek, and J.G.J. Oliver: Uncertainty in the global source distribution of nitrous oxide. *J. Geophs. Res.* (in press).

**Bowden**, W.B. and F.H. Bormann, 1986: Transport and loss of nitrous oxide in soil water after forest clear-cutting. *Science*, **233**, 867-869.

**Bronson**, K.F., A.R. Mosier, and S.R. Bishnoi, 1992: Nitrous oxide emissions in irrigated corn as affected by nitrification inhibitors. *Soil Sci. Soc. Am. J.*, **56**, 161-165.

**Buresh**, R.J., T.T. Chua, E.G. Castillo, S.P. Liboon, and D.P. Garrity, 1993: Fallow and sesbania effects on soil nitrogen dynamics in lowland rice-based cropping systems. *Agron. J.*, **85**, 316-321.

**Burke**, I.C., W.K. Lauenroth, and D.P. Coffin, 1995: Recovery of soil organic matter and N mineralization in previously cultivated shortgrass steppe. *Ecological Applications* (submitted).

**Campbell**, C.A., R.P. Zentner, H.H. Janzen, and K.E. Bowren, 1990: *Crop Rotation Studies on the Canadian Prairies*. Research Branch Agriculture Canada, Publication 1841/E, Ottawa, Canada, 133 pp.

**CAST**, 1992: *Preparing U.S. Agriculture for Global Climate Change*. Task Force Report. No. 119. P.E. Waggoner, Chair. Council for Agricultural Science and Technology, Ames, IA. 96 pp.

**Cerri**, C.C., M. Bernoux, and G.J. Blair, 1994: Carbon pools and fluxes in Brazilian natural and agricultural systems and the implications for the global $CO_2$ balance. In: *Proceedings of Int. Soil Sci. Soc., Vol. 5a*. Commission IV, Acapulco, Mexico, July 10-16, 1994, pp. 399-406.

**Cicerone**, R.J., J.D. Shetter, and C.C. Delwiche, 1983: Seasonal variation of methane flux from a California rice paddy. *J. Geophys. Res.*, **88**, 7203-7209.

**Clayton**, H., J.R.M. Arah, and K.A. Smith, 1994: Measurement of nitrous oxide emissions from fertilized grassland using closed chambers. *J. Geophys. Res.*, **99**, 16,599-607.

**Cochran**, V.L., S.F. Schlentner, and A.R. Mosier, 1995: $CH_4$ and $N_2O$ flux in subarctic agricultural soils. In: *Advances in Soil Sciences: Soil Management and Greenhouse Effect* [Lal, R., J. Kimble, E. Levine, and B. Stuart (eds.)]. CRC Press Inc., Boca Raton, FL, pp. 179-186.

**Cole**, C.V., K. Flach, J. Lee, D. Sauerbeck, and B. Stewart, 1993: Agricultural sources and sinks of carbon. *Water, Air, Soil Pollut.*, **70**, 111-122.

**Conrad**, R., 1989: Control of methane production in terrestrial ecosystems. In: *Exchange of Trace Gases Between Terrestrial Ecosystems and the Atmosphere* [Andreae, M.O. and D.S. Schimel (eds.)]. John Wiley and Sons, Chichester, UK, pp. 39-58.

**Crutzen**, P.J., 1981: Atmospheric chemical processes of the oxides of nitrogen including nitrous oxide. In: *Denitrification, Nitrification and Atmospheric Nitrous Oxide* [Delwiche, C.C. (ed.)]. Wiley, New York, NY, pp. 17-44.

**Crutzen**, P.J., I. Aselmann, and W. Seiler, 1986: Methane production by domestic animals, wild ruminants, other herbivorous fauna, and humans. *Tellus*, **38B**, 271-284.

**Crutzen**, P.J. and M.O. Andreae, 1990: Biomass burning in the tropics: impact on atmospheric chemistry and biogeochemical cycles. *Science*, **250**, 1669-1678.

**Darmstadter**, J., 1993: Climate change impacts on the energy sector and possible adjustments in the MINK region. In: *Towards an Integrated Impact Assessment of Climate Change: The MINK Study* [Rosenberg, N.J. (ed.)]. *Paper 5, Climatic Change*, **24**, 117-129.

**Davidson**, E.A. and I.L. Ackerman, 1993: Changes in soil carbon inventories following cultivation of previously untilled soils. *Biogeochemistry*, **20**, 161-164.

**Delmas**, R., 1993: An overview of present knowledge of methane emission from biomass burning. In: *Methane and Nitrous Oxide Methods in National Emission Inventories and Options for Control* [van Amstel, A.R. (ed.)]. RIVM Report No. 481507003, National Institute of Public Health and Environmental Protection, Bilthoven, Netherlands, pp. 171-185.

**Detwiler**, R.P., 1986: Land use change and the global carbon cycle: the role of tropical soils. *Biogeochemistry*, **2**, 67-93.

**Dick**, W.A., D.M. Van Doren, Jr., G.B. Triplett, Jr., and J.E. Henry, 1986: *Influence of Long-Term Tillage and Rotation Combinations on Crop Yields and Selected Soil Parameters. II. Results Obtained for a Typic Fragiudalf Soil*. Research Bulletin 1181, The Ohio State University, Ohio Agricultural Research and Development Center, Wooster, OH, 34 pp.

**Dobbie**, K.E. and K.A. Smith, 1994: Effect of land use on the rate of uptake of methane by surface soils in Northern Europe. *Annales Geophysicae,* **Supplement II, Part II**, 12:C388.

**Doerge**, T.A., R.L. Roth, and B.R. Gardner, 1991: *Nitrogen Fertilizer Management in Arizona*. College of Agriculture, University of Arizona, Tucson, AZ, 87 pp.

**Donigian, Jr**., A.S., T.O. Barnwell, Jr., R.B. Jackson IV, A.S. Patwardhan, K.B. Weinrich, A.L. Rowell, R.V. Chinnaswamy, and C.V. Cole, 1994. *Assessment of Alternative Management Practices and Policies Affecting Soil Carbon in Agroecosystems of the Central United States*. Report EPA/600/4-94/067, U.S. EPA, Athens, GA, 194 pp.

**Dormaar**, J.F. and S. Smoliak, 1985: Recovery of vegetative cover and soil organic matter during revegetation of abandoned farmland in a semi-arid climate. *Journal of Range Management*, **38**, 487-491.

**Dorr**, H., L. Katruff, and I. Levin, 1993: Soil texture parameterization of the methane uptake in aerated soils. *Chemosphere*, **26**, 697-713.

**Duxbury**, J.M., D.R. Bouldin, R.E. Terry, and R.L. Tate III, 1982: Emissions of nitrous oxide from soils. *Nature*, **298**, 462-464.

**Duxbury**, J.M. and P.K. McConnaughey, 1986: Effect of fertilizer source on denitrification and nitrous oxide emissions in a maize field. *Soil Sci. Soc. Am. J.,* **50**, 644-648.

**Duxbury**, J.M. and A.R. Mosier, 1993: Status and issues concerning agricultural emissions of greenhouse gases. In: *Agricultural Dimensions of Global Climate Change* [Kaiser, H.M. and T.E. Drennen (eds.)]. St. Louis Press. Delray Beach, FL, pp. 229-258.

**Duxbury**, J.M., L.A. Harper, and A.R. Mosier, 1993: Contributions of agroecosystems to global climate change. In: *Agricultural Ecosystem Effects on Trace Gases and Global Climate Change* [Harper, L.A., A.R. Mosier, J.M. Duxbury, and D.E. Rolston (eds.)]. ASA Special Publication Number 55, Madison, WI, pp. 1-18.

**Eichner**, M.J., 1990: Nitrous oxide emissions from fertilized soils: summary of available data. *J. Environ. Qual.,* **19**, 272-280.

**Elwell**, H.A., 1993: Development and adoption of conservation tillage practices in Zimbabwe. In: *Soil Tillage in Africa: Needs and Challenges*. Chapter 10, FAO Soils Bulletin 69, Rome, Italy, pp. 129-164.

**Enquete Commision**, 1995: *Protecting Our Green Earth. How to Manage Global Warming through Environmentally Sound Farming and Preservation of the World's Forests*. Economica Verlag, Bonn, Germany, 683 pp.

**Eswaran**, H., E. van den Berg, and P. Reich, 1993: Organic carbon in soils of the world. *Soil Sci. Soc. Am. J.,* **57**, 192-194.

**FAO**, 1990a: *Fertilizer Yearbook*. Volume 39. FAO Statistics Series No. 95. FAO, Rome, Italy, 127 pp.

**FAO**, 1990b: *Production Yearbook*. Volume 43. FAO Statistics Series No. 94. FAO, Rome, Italy, 346 pp.

**FAO**, 1990c: *The State of Food and Agriculture 1990*. David Lubin Memorial Library Cataloging in Publication Data, FAO, Rome, Italy.

**Fisher**, M.J., I.M. Rao, M.A. Ayarza, C.E. Lascano, J.I. Sanz, R.J. Thomas, and R.R. Vera, 1994: Carbon storage by introduced deep rooted grasses in the South American savannas. *Nature*, **371**, 236-238.

**Flaig**, H. and H. Mohr (eds.), 1994: *Energie aus Biomasse - eine Chance fuer die Landwirtschaft*. Springer, Berlin-Heidelberg-New York, 376 pp.

**Follett**, R.F., 1993: Global climate change, U.S. agriculture, and carbon dioxide. *J. Prod. Agric.*, **6**, 181-190.

**Frye**, W.W., 1984: Energy requirement in no-tillage. In: *No Tillage Agricultural Principles and Practices* [Phillips, R.E. and S.H. Phillips (eds.)]. Van Nostrand Reinhold, New York, NY, pp. 127-151.

**Gaedeken**, D., D. Rath, and D. Sauerbeck, 1990: Methane production in ruminants. In: *USEPA-IPCC Proc. Workshop on Greenhouse Gas Emissions from Agricultural Systems, Vol. 2*. USEPA-OPA (PM221), Washington, DC, 9 pp.

**Galbally**, I.E., P.J. Fraser, C.P. Meyer, and D.W.T. Griffith, 1992: Biosphere-atmosphere exchange of trace gases over Australia. In: *Australia's Renewable Resources: Sustainability and Global Change* [Gifford, R.M. and M.M. Barson (eds.)]. Bureau of Rural Resources, Proceedings No. 14, P.J. Grills, Commonwealth Printer, Canberra, Australia, pp. 117-149.

**Gaston**, G.G., T. Kolchugina, T.S. Vinson, 1993: Potential effect of no-till management on carbon in the agricultural soils of the former Soviet Union. *Agric., Ecosyst., Environ.*, **45**, 295-309.

**Gebhart**, D.L., H.B. Johnson, H.S. Mayeux, and H.W. Polley: The CRP increases soil organic carbon. *J. Soil and Water Conserv*. (in press).

**Gibbs**, M.J. and R.A. Leng, 1993: Methane emissions from livestock. In: *Methane and Nitrous Oxide* [van Amstel, A.R. (ed.)]. RIVM Report No. 481507003, National Institute of Public Health and Environmental Protection, Bilthoven, Netherlands, pp. 73-79.

**Gibbs**, M.J. and J.W. Woodbury, 1993: Methane emissions from livestock manure. In: *Methane and Nitrous Oxide* [van Amstel, A.R. (ed.)]. RIVM Report No. 481507003, National Institute of Public Health and Environmental Protection, Bilthoven, The Netherlands, pp. 81-91.

**Goldemberg**, J., L.C. Monaco, and I.C. Macedo, 1993: The Brazilian fuel-alcohol program. In: *Renewable Energy: Sources for Fuels and Electricity* [Burnham, L. (exe. ed.)]. Island Press, Washington, DC, pp. 841-863.

**Graef**, M., G. Vellguth, J. Krahl, and A. Munack, 1994: Fuel from sugar beet and rape seed oil—mass and energy balances for evaluation. Proceedings of the 8th European Conference on Biomass for Energy, Environment, Agriculture and Industry, 3-5 October 1994, Vienna, Austria, pp. 1-4.

**Graham**, R.L., L.L. Wright, and A. Turhollow, 1993: The potential for short rotation woody crops to reduce $CO_2$ emissions. *Climatic Change*, **22**, 223-238.

**Granli**, T. and O.C. Bockman, 1994: Nitrous oxide from agriculture. *Norwegian Journal of Agricultural Sciences,* **Supplement No. 12**, 128.

**Greenland**, D.J. and P.H. Nye, 1959: Increases in carbon and nitrogen contents of tropical soils under natural fallows. *J. Soil Sci*., **10**, 284-298.

**Gupta**, R.K. and D.L.N. Rao, 1994: Potential of wastelands for sequestering carbon by reforestation. *Current Science*, **66**, 378-380.

**Haas**, H.J., C.E. Evans, and E.F. Miles, 1957: *Nitrogen and Carbon Changes in Great Plains Soils as Influenced by Cropping and Soil Treatments*. Technical Bulletin No. 1164 USDA, State Agricultural Experiment Stations, Washington, DC, 111 pp.

**Haas**, G., U. Geier, D.G. Schulz, and U. Koepke, 1995a: Klimarelevanz des Agrarsektors der Bundesrepublik Deutschland: Reduzierung der Emission von Kohlendioxid. Berichte ueber Landwirtschaft 73, Landwirtschaftsverlag GmbH, Muenster-Hiltrup, Germany, pp. 387-400.

**Haas**, G., U. Geier, D.G. Schulz, and U. Koepke, 1995b: Vergleich Konventioneller und Organischer Landbau—Teil I: Klimarelevante Kohlendioxid-Emission durch den Verbrauch fossiler Energie. Berichte ueber Landwirtschaft 73, Landwirtschaftsverlag GmbH, Muenster-Hiltrup, Germany, pp. 401-415.

**Hall**, D.O., F. Rosillo-Calle, R.H. Williams, and J. Woods, 1993: Biomass for energy supply prospects. In: *Renewables for Fuels and Electricity* [Johansson, B.J., H. Kelly, A.K.N. Reddy, and R.H. Williams (eds.)]. Island Press, Washington, DC, pp. 583-652.

**Hao**, W.M., D. Scharffe, and P.J. Crutzen, 1988: Production of $N_2O$, $CH_4$, and $CO_2$ from soils in the tropical savanna during the dry season. *J.Atmos Chem*., **7**, 93-105.

**Hao**, W.M., M.H. Liu, and P.J. Crutzen, 1990: Estimates of annual and regional releases of $CO_2$ and other trace gases to the atmosphere from fires in the tropics, based on the FAO statistics for the period 1975-1980. In: *Fire in the Tropical Biota* [Goldhammer, J.G. (ed.)]. Springer Verlag, Berlin, Germany. *Ecological Studies*, **84**, 440-462.

**Haynes**, R.J., R.S. Swift, and R.C. Stephen, 1991: Influence of mixed cropping rotations (pasture-arable) on organic matter content, water stable aggregation and clod porosity in a group of soils. *Soil & Tillage Research*, **19**, 77-81.

**Hogan**, K.B., 1993: Methane reductions are a cost-effective approach for reducing emissions of greenhouse gases. In: *Methane and Nitrous Oxide: Methods in National Emissions Inventories and Options for Control* [van Amstel, A.R. (ed.)]. RIVM Report No. 481507003, Bilthoven, Netherlands, pp. 187-201.

**Houghton**, R.A., J.E. Hobbie, J.M. Melillo, B. Moore, B.J. Peterson, G.R. Shaver, and G.M. Woodwell, 1983: Changes in the carbon content of terrestrial biota and soils between 1860 and 1980: a net release of $CO_2$ to the atmosphere. *Ecol. Monogr*., **53**, 235-262.

**Houghton**, R.A. and D.L. Skole, 1990: Carbon. In: *The Earth as Transformed by Human Action* [Turner, B.L., W.C. Clark, R.W. Kates, J.F. Richards, J.T. Mathews, and W.B. Meyer (eds.)]. Cambridge University Press, New York, NY, pp. 393-408.

**Houghton**, R.A., 1994: The worldwide extent of land-use change. *BioScience*, **44**, 305-313.

**Hütsch**, B.W., C.P. Webster, and D.S. Powlson, 1993: Long-term effects of nitrogen fertilization on methane oxidation in soil of the broadbalk wheat experiment. *Soil Biol. Biochem*., **25**, 1307-1315.

**Hurst**, D.F., D.W.T. Griffith, J.N. Carras, D.S. Williams, and P.J. Fraser, 1994: Measurement of trace gases emitted by Australian savanna fires during the 1990 dry season. *J. Atmos. Chem.,* **18**, 33-56.

**IPCC**, 1990a: *Climate Change: The IPCC Scientific Assessment*. IPCC Working Group I [Houghton, J.T., G.J. Jenkins, and J.J. Ephraums (eds.)] and WMO/UNEP. Cambridge University Press, Cambridge, UK, 365 pp.

**IPCC**, 1990b: *Climate Change: The IPCC Impacts Assessment*. IPCC Working Group II [McG. Tegart, W.J., G.W. Sheldon, and D.C. Griffiths (eds.)] and WMO/UNEP. Australian Government Publishing Service, Canberra, Australia, 275 pp.

**IPCC**, 1991: *Climate Change: The IPCC Response Strategies*. IPCC Working Group III and WMO/UNEP. Island Press, Covelo, CA, 273 pp.

**IPCC**, 1992: *Climate Change 1992: The IPCC Supplementary Report to the IPCC Scientific Assessment*. Supporting Material, Working Group III, Response Strategies, Subgroup AFOS, WMO/UNEP, Geneva, pp. 14-22.

**IPCC**, 1994: *Radiative Forcing of Climate Change*. The 1994 Report of the Scientific Assessment Working Group of IPCC, Summary for Policymakers. WMO/UNEP, Geneva, Switzerland, 28 pp.

**Isermann**, K., 1993: Territorial, continental and global aspects of C, N, P and S emissions from agricultural ecosystems. In: *NATO Advanced Research Workshop (ARW) on Interactions of C,N, P and S Biochemical Cycles*. Springer-Verlag. Heidelberg, Germany, ASI Series, pp. 79-121.

**Isermann**, K., 1994a: Agriculture's share in the emission of trace gases affecting the climate and some cause-oriented proposals for sufficiently reducing this share. *Environ. Pollut*., **83**, 95-111.

**Isermann**, K., 1994b: *Ammoniak*. Enquete-Kommission "Schutz der Erdatmosphaere" (ed.) Landwirtschaft, Studienprogramm, Economica Verlag, Bonn, Germany.

**Isermann**, K. and R. Isermann, 1995: Present situation, demands, possible solutions and outlooks for a sustainable agriculture, human nutrition, waste and wastewater management before the background of nutrient balances within the EU [Barrage, A. and X. Edelmann (eds.)]. R'95 Congress (Recovery/Recycling/Re-Integration) Proc. IV, Eidgen, Materialpruefungs - u. Forschungsanstalt (EMPA), Duebendorf, Switzerland, pp. 151-156.

**Ismail**, I., R.L. Blevins, and W.W. Frye, 1994: Long-term no-tillage effects on soil properties and continuous corn yields. *Soil Sci. Soc. Am. J*., **58**, 193-196.

**Janzen**, H.H., 1987: Soil organic matter characteristics after long-term cropping to various spring wheat rotations. *Can. J. Soil Sci*., **67**, 845.

**Jenkinson**, D.S., 1971: The accumulation of organic matter in soil left uncultivated. Rothamsted Exp. Sta. Ann. Rep. for 1970, part 2, pp. 113-137.

**Jenkinson**, D.S., D.D. Harkness, E.D. Vance, D.E. Adams, and A.F. Harrison, 1992: Calculating net primary production and annual input of organic matter to soil from the amount and radiocarbon content of soil organic matter. *Soil Biology and Biochemistry,* **24**, 295-308.

**Johansson**, B.J., H. Kelly, A.K.N. Reddy, and R.H. Williams (eds.), 1993: *Renewables for Fuels and Electricity*. Island Press, Washington, DC, 1160 pp.

**Johnson**, D.E., G.M. Ward, and J. Torrent, 1991: The environmental impact of the use of bST in dairy cattle. *J. Environ. Qual*., **21**, 157-162.

**Johnson**, D.E., T.M. Hill, G.M. Ward, K.A. Johnson, M.E. Branine, B.R. Carmean, and D.W. Lodman, 1993: Ruminants and other animals. In: *Atmospheric Methane: Sources, Sinks, and Role in Global Change* [Khalil, M.A.K. (ed.)]. Springer-Verlag, New York, NY, pp. 199-229.

**Kaltschmitt**, M. and S. Becher, 1994: Biomassenutzung in Deutschland—Stand und Perspektiven. In: *Thermische Nutzung von Biomasses* [BMELF (ed.)]. Schriftenreihe "Nachwachsende Rohstoffe," Vol. 2, Landwirtschaftsverlag GmbH, Muenster-Hiltrup, Germany, pp. 9-25.

**Keerthisinghe**, D.G., J.R. Freney, and A.R. Mosier, 1993: Effect of wax-coated calcium carbide and nitrapyrin on nitrogen loss and methane emission from dry-seeded flooded rice. *Biol. Fertil. Soils*, **16**, 71-75.

**Keller**, M., M.E. Mitre, and R.F. Stallard, 1990: Consumption of atmospheric methane in soils of central Panama: effects of agricultural development. *Global Biogeochem. Cycles*, **4**, 21-27.

**Keller**, M., M.E. Veldkamp, A.M. Weltz, and W.A. Reiners, 1993: Effect of pasture age on soil trace-gas emissions from a deforested area of Costa Rica. *Nature*, **365**, 244-246.

**Kern**, J.S. and M.G. Johnson, 1993: Conservation tillage impacts on national soil and atmospheric carbon levels. *Soil Sci. Soc. Am. J*., **57**, 200-210.

**Kleinhanss**, W., 1993: Pflanzenoele als Treibstoff—Erzeugung, Nutzung, Perspektiven. In: *Energie aus Biomasse—eine Chance fuer die Landwirtschaft* [Flaig, H. and H. Mohr (eds.)]. Springer, Berlin-Heidelberg-New York, pp. 67-83.

**Koepf**, H., S. Kaffka, and F. Sattler, 1988: *Naehrstoffbilanz und Energiebedarf im landwirtschaftlichen Betriebsorganismus*. Verl. Freies Geistesleben, Stuttgart, Germany, 62 pp.

**Kroeze**, C., 1994: Anthropogenic emissions of nitrous oxide ($N_2O$) from Europe. *Sci. Total Environ.,* **152**, 189-205.

**Lal**, R., 1986: Soil surface management in the tropics for intensive land use and high and sustained production. *Adv. Soil Sci*., Vol. 5, Springer-Verlag, Inc., New York, NY, 109 pp.

**Lal**, R. and T.J. Logan: Agricultural activities and greenhouse gas emissions from soils of the tropics. In: *Soil Management and Greenhouse Effect* [Lal, R. *et al*. (eds.)]. Lewis Publishers, Chelsea, MI (in press).

**Lauren**, J.G. and J.M. Duxbury, 1993: Methane emissions from flooded rice amended with green manure. In: *Agroecosystem Effects on Radiatively Important Trace Gases and Global Climate Change* [Harper, L.A., A.R. Mosier, J.M. Duxbury, and D.E. Rolston (eds.)]. ASA Special Publication No. 55, Madison, WI, pp. 183-192.

**Lauren**, J.G., G.S. Pettygrove, and J.M. Duxbury, 1994: Methane emissions associated with a green manure amendment to flooded rice in California. *Biogeochem*., **24**, 53-65.

**Lee**, J.J., D.L. Phillips, and R. Liu, 1993: The effect of trends in tillage practices on erosion and carbon content of soils in the US corn belt. *Water, Air, Soil Pollut*., **70**, 389-401.

**Leible**, L. and D. Wintzer, 1993: Energiebilanzen bei nach-wachsenden Energietraegern. In: *Energie aus Biomasse - eine Chance fuer die Landwirtschaft* [Flaig, H. and H. Mohr (eds.)]. Springer, Berlin-Heidelberg-New York, pp. 67-83.

**Leng**, R.A., 1991: *Improving Ruminant Production and Reducing Methane Emissions from Ruminants by Strategic Supplementation*. USEPA report 400/1-91/004, Office of Air and Radiation, Washington, DC.

**Levine**, J.S., W.R. Cofer, and J.P. Pinto, 1993: Biomass burning. In: *Atmospheric Methane: Sources, Sinks and Role in Global Change* [Khalil, M.A.K. (ed.)]. Springer-Verlag, Berlin, Heidelberg, Germany, pp. 230-253.

**Lin**, E., H. Dong, and Y. Li, 1994: Methane emissions of China: agricultural sources and mitigation options. In: *Non-$CO_2$ Greenhouse Gases* [van Ham, J. *et al*. (eds.)]. Kluwer Academic Publishers, Dordrecht, Netherlands, pp. 405-410.

**Lindau**, C.W., P.K. Bollich, R.D. DeLaune, A.R. Mosier, and K.F. Bronson, 1993: Methane mitigation in flooded Louisiana rice fields. *Biol. Fertil. Soils*, **15**, 174-178.

**Lobert**, J.M., D.H. Scharffe, W.M. Hao, and P.J. Crutzen, 1990: Importance of biomass burning in the atmospheric budgets of nitrogen - containing gases. *Nature*, **346**, 552-554.

**Lugo**, A.E. and S. Brown, 1993: Management of tropical soils as sinks or sources of atmospheric carbon. *Plant and Soil*, **149**, 27-41.

**Mann**, L.K., 1986: Changes in soil carbon storage after cultivation. *Soil Sci*., **142**, 279-288.

**Marland**, G. and A.F. Turhollow, 1991: $CO_2$ emissions from the production and combustion of fuel ethanol from corn. *Energy*, **16**, 1307-1316.

**McTaggart**, I., H. Clayton, and K. Smith, 1994: Nitrous oxide flux from fertilized grassland: strategies for reducing emissions. In: *Non-$CO_2$ Greenhouse Gases* [van Ham, J. *et al*. (eds.)]. Kluwer Academic Publishers, Dordrecht, Netherlands, pp. 421-426.

**Minami**, K. and A. Ohsawa, 1990: Emission of nitrous oxide dissolved in drainage water from agricultural land. In: *Soils and the Greenhouse Effect* [Bouwman, A.F. (ed.)]. John Wiley and Sons, New York, NY, pp. 503-509.

**Minami**, K., T. Shibuya, Y. Ogawa, and S. Fukushi, 1990: Effect of nitrification inhibitors on emission of nitrous oxide from soils. *Trans 14th Int. Congr. Soil Sci*., **2**, 267-272.

**Minami**, K., J. Goudriaan, E.A. Lantinga, and T. Kimura, 1993: The significance of grasslands in emission and absorption of greenhouse gases. Proc. 17th Int. Grassland Congr., pp. 1231-1238.

**Minami**, K., 1993: Methane from rice production. In: *Methane and Nitrous Oxide* [van Amstel, A.R. (ed.)]. Proceedings IPCC Workshop, Bilthoven, Netherlands, pp. 143-162.

**Minami**, K., 1994: Effect of nitrification inhibitors and slow-release fertilizer on emission of nitrous oxide from fertilized soils, In: *$CH_4$ and $N_2O$: Global Emissions and Controls from Rice Fields and Other Agricultural and Industrial Sources* [Minami, K., A. Mosier, and R. Sass (eds.)]. NIAES Series 2. Yokendo Publishers, Tokyo, Japan, pp. 187-196.

**Minami**, K., A. Mosier, and R. Sass (eds.), 1994: *$CH_4$ and $N_2O$: Global Emissions and Controls from Rice Fields and Other Agricultural and Industrial Sources*. NIAES Series 2, Yokendo Publishers, Tokyo, Japan, 234 pp.

**Mosier**, A.R., D.S. Schimel, D. Valentine, K.F. Bronson, and W.J. Parton, 1991: Methane and nitrous oxide fluxes in native, fertilized, and cultivated grasslands. *Nature*, **350**, 330-332.

**Mosier**, A.R., 1993: Nitrous oxide emissions from agricultural soils. In: *Methane and Nitrous Oxide: Methods in National Emission Inventories and Options for Control Proceedings* [van Amstel, A.R. (ed.)]. National Institute of Public Health and Environmental Protection, Bilthoven, Netherlands, pp. 273-285.

**Mosier**, A.R. and A.F. Bouwman, 1993: Working group report: nitrous oxide emissions from agricultural soils. In: *Methane and Nitrous Oxide: Methods in National Emission Inventories and Options for Control Proceedings* [van Amstel, A.R. (ed.)]. National Institute of Public Health and Environmental Protection, Bilthoven, Netherlands, pp. 343-346.

**Mosier**, A.R., J.M. Duxbury, J.R. Freney, O. Heinemeyer, and K. Minami, 1995: Nitrous oxide emission from agricultural fields: assessment, measurements and mitigation. *Plant and Soil* (in press).

**Nesbit**, S.P. and G.A. Breitenbeck, 1992: A laboratory study of factors influencing methane uptake by soils. *Agric. Ecosyst. Environ*., **41**, 39-54.

**Neue**, H.U., 1992: Agronomic practices affecting methane fluxes from rice cultivation. In: *Trace Gas Exchange in a Global Perspective* [Ojima, D.S. and B.H. Svensson (eds.)]. *Ecol. Bull., Copenhagen*, **42**, 174-182.

**Neue**, H.U., R.S. Lantin, R. Wassmann, J.B. Aduna, M.C.R. Alberto, and M.J.F. Andales, 1994: Methane emission from rice soils of the Philippines. In: *$CH_4$ and $N_2O$: Global Emissions and Controls from Rice Fields and Other Agricultural and Industrial Sources* [Minami, K., A. Mosier, and R. Sass (eds.). NIAES, Yokendo Publishers, Tokyo, Japan, pp. 55-63.

**Nguyen**, M. L. and R.J. Haynes, 1995: Energy and labour efficiency for three pairs of conventional and alternative mixed cropping (pasture-arable) farms in Canterbury, New Zealand. *Agricultue, Ecosystems and Environment*, **52**, 163-172.

**Nouchi**, I., S. Mariko, and K. Aoki, 1990: Mechanisms of methane transport from the rhizosphere to the atmosphere through rice plant. *Plant Physiol*., **94**, 59-66.

**Nouchi**, I., 1994a: Mechanisms of methane transport through rice plants. In: *$CH_4$ and $N_2O$: Global Emissions and Controls from Rice Fields and Other Agricultural and Industrial Sources* [Minami, K., A. Mosier, and R. Sass (eds.)]. NIAES, Yokendo Publishers, Tokyo, Japan, pp. 87-104.

**Nouchi**, I., T. Hosono, K. Aoki, and K. Minami, 1994b: Seasonal variation in methane flux from rice paddies associated with methane concentration in soil water, rice biomass and temperature, and its modeling. *Plant and Soil*, **161**, 195-208.

**OECD/OCDE**, 1991: *Estimation of Greenhouse Gas Emissions and Sinks*. Final report from the OECD Experts Meeting, 18-21 February 1991, Prepared for Intergovennmental Panel on Climate Change, Washington, DC, Revised August, 1991.

**Ojima**, D.S., D.W. Valentine, A.R. Mosier, W.J. Parton, and D.S. Schimel, 1993a: Effect of land use change on methane oxidation in temperate forest and grassland soils. *Chemosphere,* **26**, 675-685.

**Ojima**, D.S., B.O.M. Dirks, E.P. Glenn, C.E. Owensby, and J.M.O. Scurlock, 1993b: Assessment of C budget for grasslands and drylands of the world. *Water, Air, Soil Pollut*., **70**, 95-109.

**Ojima**, D.S., W.J. Parton, D.S. Schimel, J.M.O. Scurlock, and T.G.F. Kittel, 1993c: Modeling the effects of climatic and $CO_2$ changes on grassland storage of soil C. *Water, Air, Soil Pollut*., **70**, 643-657.

**Oldeman**, L.R., V.W.P. van Engelen, and J.H.M. Pulles, 1990: The extent of human-induced soil degradation. In: *World Map of the Status of Human-Induced Soil Degradation: An Expanatory Note* [Oldeman, L.R., R.T.A. Hakkeling, and W.G. Sombroek (eds.)]. International Soil Reference and Information Centre, Wageningen, Netherlands.

**Parashar**, D.C., A.P. Mitra, S.K. Sinha, P.K. Gupta, J. Rai, R.C. Sharma, N. Singh, S. Kaul, G. Lai, A. Chaudhary, H.S. Ray, S.N. Das, K.M. Parida, S.B. Rao, S.P. Kanungo, T. Ramasami, B.U. Nair, M. Swamy, G. Singh, S.K. Gupta, A.R. Singh, B.K. Saikia, A.K.S. Barua, M.G. Pathak, C.P.S. Iyer, M. Gopalakrishnan, P.V. Sane, S.N. Singh, R. Banerjee, N. Sethunathan, T.K. Adhya, V.R. Rao, P. Palit, A.K. Saha, N.N. Purkait, G.S. Chaturvedi, S.P. Sen, M. Sen, B. Arkrkar, A. Banik, B.H. Subbarary, S. Lal, and S. Venkatramani, 1994: Methane budget from Indian paddy fields. In: *$CH_4$ and $N_2O$: Global Emissions and Controls from Rice Fields and Other Agricultural and Industrial Sources* [Minami, K., A. Mosier, and R. Sass (eds.)]. NIAES. Yokendo Publishers, Tokyo, Japan, pp. 27-39.

**Parton**, W.J., D.S. Schimel, C.V. Cole, and D.S. Ojima, 1987: Analysis of factors controlling soil organic matter levels in Great Plains grasslands. *Soil Sci. Soc. Am. J.,* **51**, 1173-1179.

**Parton**, W.J., J.M.O. Scurlock, D.S. Ojima, T.B. Gilmanov, R.J. Scholes, D.S. Schimel, T.B. Kirchner, J.-C. Meanut, T. Seastedt, E. Garcia Moya, A. Kamnalrut, and J.I. Kinyamario, 1993: Observations and modeling of biomass and soil organic matter dynamics for the grassland biome worldwide. *Global Biogeochemical Cycles*, **7**, 785-809.

**Paul**, E.A., K. Paustian, E.T. Elliott, and C.V. Cole (eds.), 1995: *Soil Organic Matter in Temperate Agroecosystems. Long-Term Experiments in North America*. Lewis Publishers, Chelsea, MI (in press).

**Paustian**, K., H.P. Collins, and E.A. Paul, 1995: Management controls on soil carbon. In: *Soil Organic Matter in Temperate Agroecosystems. Long-Term Experiments of North America* [Paul, E.A., K. Paustian, E.T. Elliott, and C.V. Cole (eds.)]. CRC/Lewis Publishers, Chelsea, MI (in press).

**Peoples**, M.B., A.R. Mosier, and J.R. Freney, 1995: Minimizing gaseous loss of nitrogen. In: *Nitrogen Fertilization in the Environment* [Bacon, P.E. (ed.)]. Marcel Dekker Inc., New York, NY, pp. 565-602.

**Pimentel**, D., W. Dazhong, and M. Giampietro, 1990: Technological changes in energy use in U.S. agricultural production. In: *Agroecology: Researching the Ecological Basis for Sustainable Agriculture* [Gliessman, S.R. (ed.)]. Ecological Studies: Analysis and Synthesis (USA), **78**, 305-321, Springer, New York, NY.

**Potter**, C.S., J.T. Randerson, C.B. Field, P.A. Matson, P.M. Viousek, H.A. Mooney, and S.A. Klooster, 1993: Terrestrial ecosystem production: a process model based on global satellite and surface data. *Global Biogeochemical Cycles*, **7**, 811-842.

**Powlson**, D.S., P.C. Brookes, and B.T. Christensen, 1987: Measurement of soil microbial biomass provides an early indication of changes in total soil organic matter due to straw incorporation. *Soil Biol. Biochem.*, **19**, 159-164.

**Powlson**, D.S., P. Smith, and J.U. Smith (eds.), 1995: *Evaluation of Soil Organic Matter Models Using Existing Long-Term Data Sets*. NATO ASI Series, Springer Verlag (in press).

**Rahm**, M.R. and W.E. Huffman, 1984: The adoption of reduced tillage: the role of human capital and other variables. *Am. J. Agric. Econ.*, **66**, 405-413.

**Raich**, J.W., E.B. Rastetter, J.M. Melillo, D.W. Kicklighter, P.A. Steudler, B.J. Peterson, A.L. Grace, B. Moore III, and C.J. Vorosmarty, 1991: Potential net primary productivity in South America: application of a global model. *Ecol. Applic.*, **1**, 399-429.

**Rasmussen**, P.E. and H.P. Collins, 1991: Long-term impacts of tillage, fertilizer, and crop residue on soil organic matter in temperate semi-arid regions. *Adv. Agron.*, **45**, 93.

**Reeburgh**, W.S., S.C. Whalen, and M.J. Alpern, 1993: The role of methylotrophy in the global methane budget. In: *Microbial Growth on $C_1$ Compounds*. pp. 1-14.

**Reinhardt**, G.A., 1993: Energie- und $CO_2$-Bilanzierung nachwachsender Rohstoffe. Theoretische Grundlagen und Fallstudie Raps. Vieweg, Wiesbaden, Germany.

**Ryden**, J.C., 1981: $N_2O$ exchange between a grassland soil and the atmosphere. *Nature*, **292**, 235-237.

**Saffigna**, P.G., D.S. Powlson, P.C. Brookes, and G.A. Thomas, 1989: Influence of sorghum residues and tillage on soil organic matter and soil microbial biomass in an Australian Vertisoil. *Soil Biol. Biochem.*, **21**, 759-765.

**Safley**, L.M., M.E. Casada, J.W. Woodbury, and K.F. Roos, 1992: *Global Methane Emissions from Livestock and Poultry Manure*. USEPA Report 400/1-91/048, Office of Air and Radiation, Washington, DC, 68 pp.

**Sampson**, R.N., L.L. Wright, J.K. Winjum, J.D. Kinsman, J. Benneman, E. Kursten, and J.M.O. Scurlock, 1993: Biomass management and energy. In: *Terrestrial Biospheric Carbon Fluxes: Quantification of Sinks and Sources of $CO_2$* [Wisniewski, J. and R.N. Sampson (eds.)]. Kluwer Academic Publishers, Dordrecht, Netherlands, pp. 139-162.

**Sanchez**, P.A., C.A. Palm, and T.J. Smyth, 1990: Approaches to mitigate tropical deforestation by sustainable soil management practices In: *Soils on a Warmer Earth* [Scharpenseel, H.W., M. Schomaker, and A. Ayoub (eds.)]. Elsevier, Amsterdam, Netherlands, pp. 211-220.

**Sass**, R.L., 1994: Short summary chapter for methane. In: *$CH_4$ and $N_2O$: Global Emissions and Controls from Rice Fields and Other Agricultural and Industrial Sources* [Minami, K., A. Mosier, and R. Sass (eds.)]. NIAES, Yokendo Publishers, Tokyo, Japan, pp. 1-7.

**Sass**, R.L. and F.M. Fisher, 1994: $CH_4$ emission from paddy fields in the United States gulf coast area. In: *$CH_4$ and $N_2O$: Global Emissions and Controls from Rice Fields and Other Agricultural and Industrial Sources* [Minami, K., A. Mosier, and R. Sass (eds.)]. NIAES, Yokendo Publishers, Tokyo, Japan, pp. 65-77.

**Sass**, R.L., F.M. Fisher, F.T. Turner, and M.F. Jund, 1991: Methane emissions from rice fields as influenced by solar radiation, temperature, and straw incorporation. *Global Biogeochem. Cycles*, **5**, 335-350.

**Sass**, R.L., F.M. Fisher, Y.B. Wang, F.T. Turner, and M.F. Jud, 1992: Methane emission from rice fields: the effect of floodwater management. *Global Biogeochem. Cycles*, **6**, 249-262.

**Sauerbeck**, D., 1993: $CO_2$ emissions from agriculture: sources and mitigation potentials. *Water, Air, Soil Pollut.,* **70**, 381-388.

**Sauerbeck**, D., 1994a: Die Landwirtschaft als Verursacherin und Betroffene moeglicher Klimaveraenderungen. Klimaforschung in Bayern. In: *Rundgespraeche der Kommission fuer Oekologie* [Bayrische Akademie der Wissenschaften (ed.)]. Bd. 8, Verlg. F. Pfeil, Muenchen, pp. 151-168.

**Sauerbeck**, D., 1994b: Nitrogen fertilization and nitrogen balance—environmental consequences and limitations. In: *The Terrestrial Nitrogen Cycle as Influenced by Man* [Mohr, H.U. and K. Muentz (ed.)]. *Nova Acta Leopoldina (Halle),* **70**, 441-447.

**Schutz**, H., A. Holzapfel-Pschorn, R. Conrad, H. Rennenberg, and W. Seiler, 1989: A 3-year continuous record on the influence of daytime, season and fertilizer treatment on methane emission rates from an Italian rice paddy. *J. Geophys. Res.*, **94**, 16,405-16.

**Schwab**, A.W., M.O. Bagby, and B. Freeman, 1987: Preparation and properties of diesel fuels from vegetable oils. *Fuels,* **66(10)**, 1372-1378.

**Scurlock**, J.M.O., D.O. Hall, J.I. House, and R. Howes, 1993: Utilizing biomass crops as an energy source: a European perspective. In: *Terrestrial Biospheric Carbon Fluxes: Quantification of Sinks and Sources of $CO_2$* [Wisniewski, J. and R.N. Sampson (eds.)]. Kluwer Academic Publishers, Dordrecht, Netherlands, pp. 499-518.

**Seiler**, W., R. Conrad, and D. Scharffe, 1984: Field studies of methane emission from termite nests into the atmosphere and measurements of methane uptake by tropical soils. *J. Atmos. Chem.,* **1**, 171-186.

**Sharpley**, A.N. and J.R. Williams, 1990: *EPIC—Erosion/Productivity Impact Calculator: 1. Model Documentation*. U.S. Department of Agriculture Technical Bulletin No. 1768, Washington, DC, 235 pp.

**Shibata**, M., 1994: Methane production in ruminants. In: *$CH_4$ and $N_2O$: Global Emissions and Controls from Rice Fields and Other Agricultural and Industrial Sources* [Minami, K., A. Mosier, and R. Sass (eds.)]. NIAES, Yokendo Publishers, Tokyo, Japan, pp. 105-115.

**Sims**, R.E.H., 1990: Tallow esters and vegetable oil as alternative diesel fuels—a review of the New Zealand programme. *Solar and Wind Technology,* **7(1)**, 31-36.

**Smith**, K.A., 1990: Greenhouse gas fluxes between land surfaces and the atmosphere. *Progress in Phys. Geogr.*, **14**, 349-372.

**Smith**, K.A., G.P. Robertson, and J.M. Melillo, 1994: Exchange of trace gases between the terrestrial biosphere and the atmosphere in the mid latitudes. In: *Global Atmospheric–Biospheric Chemistry* [Prinn, R.B. (ed.)]. Plenum Press, New York, NY, pp. 179-203.

**Sombroek**, W.G., F.O. Nachtergaele, and A. Hebel, 1993: Amounts, dynamics and sequestering of carbon in tropical and subtropical soils. *Ambio*, **22**, 417-426.

**Steudler**, P.A., R.D. Bowden, J.M. Melillo, and J.D. Aber, 1989: Influence of nitrogen fertilization on methane uptake in temperate forest soils. *Nature*, **341**, 314-316.

**Swift**, M.J. and P.A. Sanchez, 1984: Biological management of tropical soil fertility for sustained productivity. *Nature Res.*, **20**, 2-10.

**Tice**, T.F. and F.M. Epplin, 1984: Cost-sharing to promote use of conservation tillage. *J. Soil Water Conserv.,* **39**, 395-397.

**Tyson**, K.C., D.H. Roberts, C.R. Clement, and E.A. Garwood, 1990: Comparison of crop yields and soil conditions during 30 years under annual tillage or grazed pasture. *J. Agric. Sci.*, **115**, 29-40.

**U.S. EPA**, 1990a: *Greenhouse Gas Emissions from Agricultural Systems*. Proc. IPCC-RSWG-AFOS Workshop, Vol. 1 and 2, USEPA-OPA, Washington, DC, IPCC/WMO, Geneva, Switzerland.

**U.S. EPA**, 1990b: *Policy Options for Stabilizing Global Climate. Draft Report to Congress* [Tirpak, D. and D. Lashof (eds.)]. U.S. Environmental Protection Agency, Washington, DC, 810 pp.

**U.S. EPA**, 1992: *Anthropogenic Methane Emissions in the US: Estimates for 1990*. Report No. 430-R-93-003, U.S. Environmental Protection Agency, Office of Air and Radiation, Washington, DC.

**U.S. EPA**, 1993: Methane emissions from livestock manure. In: *Global Methane Emissions Report to Congress*. Prepared by Climate Change Division, Office of Policy, Planning and Evaluation, Environmental Protection Agency, Washington, DC, Review Draft.

**U.S. EPA**, 1994: *International Anthropogenic Methane Emissions: Estimates for 1990*. EPA 20-R-93-010, U.S. Environmental Protection Agency, Office if Policy, Planning and Evaluation, Washington, DC, 2122 pp.

**U.S. Government Executive Office**, 1995: *Economic Report of the President*. U.S. Government Printing Office, Washington, DC, 407 pp.

**van Heerwarden**, K., 1992: Agriculture and the greenhouse effect. *Change*, **II**, 17-20.

**Vellguth**, G., 1983: Performance of vegetable oils and their monoesters as fuels for diesel engines. Proceedings of the International Off-Highway Meeting and Exposition, Milwaukee, WI, September 12-15, Society of Automotive Engineers, pp. 1-10.

**Wang**, Mingxing, Dai Aiguo, Shangguan Xingjian, Ren Lixin, Shen Renxing, H. Schutz, W. Seiler, R.A. Rasmussen, and M.A.K. Khalil, 1994: Sources of methane in China. In: *$CH_4$ and $N_2O$: Global Emissions and Controls from Rice Fields and Other Agricultural and Industrial Sources* [Minami, K., A. Mosier, and R. Sass (eds.)]. NIAES, Yokendo Publishers, Tokyo, Japan, pp. 9-26.

**Watson**, R.T., H. Rodhe, H. Oeschger, and U. Siegenthaler, 1990: Greenhouse gases and aerosols. In: *Climate Change: The IPCC Scientific Assessment* [Houghton, J.T., G.J. Jenkins, and J.J. Ephraums (eds.)]. Cambridge University Press, Cambridge, UK, pp. 1-40.

**Watson**, R.T., L.G. Meira Filho, E. Sanhueza, and T. Janetos, 1992: Greenhouse gases: sources and sinks. In: *Climate Change 1992: The Supplementary Reports to the IPCC Scientific Assessment* [Houghton, J.T., B.A. Callander, and S.K. Varney (eds.)]. Cambridge University Press, Cambridge, UK, pp. 25-46.

**Wood**, B.J. and R.H.V. Conley, 1991: The energy balance of oil palm cultivation. Proceedings 1991 PORIM Int. Palm Oil Conference, pp. 130-143.

**Wright**, L.L. and E. Hughes, 1993: U.S. carbon offset potential using biomass energy systems. In: *Terrestrial Biospheric Carbon Fluxes: Quantification of Sinks and Sources of $CO_2$*, [Wisniewski, J. and R.N. Sampson (eds.)]. Kluwer Academic Publishers, Dordrecht, Netherlands, pp. 483-498.

**Yagi**, K. and K. Minami, 1990: Effect of organic matter application on methane emission from some Japanese paddy fields. *Soil Sci. Plant Nutr.*, **36**, 599-610.

**Yagi**, K. and K. Minami, 1993: Spatial and temporal variations of methane flux from a rice paddy field. *Biogeochemistry of Global Change* [Oremland, R.S. (ed.)]. Chapman and Hall, New York, NY, pp. 353-368.

**Yagi**, K., H. Tsuruta, K. Mianmi, P. Chairoj, and W. Cholitkul, 1994: Methane emission from Japanese and Thai paddy fields. In: *$CH_4$ and $N_2O$: Global Emissions and Controls from Rice Fields and Other Agricultural and Industrial Sources* [Minami, K., A. Mosier, and R. Sass (eds.)]. NIAES, Yokendo Publishers, Tokyo, Japan, pp. 41-53.