Climate Change and Global Crop Productivity

Edited by K.R. Reddy and H.F. Hodges, Department of Plant and Soil Sciences, Mississippi State University, USA

Current and predicted worldwide climatic changes have raised concerns about potential crop yields and production systems. Such concerns include the ability to accommodate these uncertain effects in order to ensure an adequate food supply for an increasing population. This book is the first comprehensive examination of the potential effects climate change will have on crop production systems. It also reviews the effects such systems have on climate change itself. There are individual chapters on the main cereal crops, soybean, cotton, vegetables, roots and tubers, as well as on grasslands, trees and rangelands. Environmental factors influence a number of plant physiological processes uniquely. These chapters discuss the mechanisms of species responses to temperature, carbon dioxide, radiation, water and nutrients. Playing an important role in fostering dialogue among the scientific community, including policy makers, and in furthering integrated responses to global climatic change, this book is written by leading international authorities from the USA, Europe, Japan, Australia and New Zealand. It is indispensable for advanced students and researchers in crop science, including breeding and technology, environmental plant physiology, ecology and climatology.

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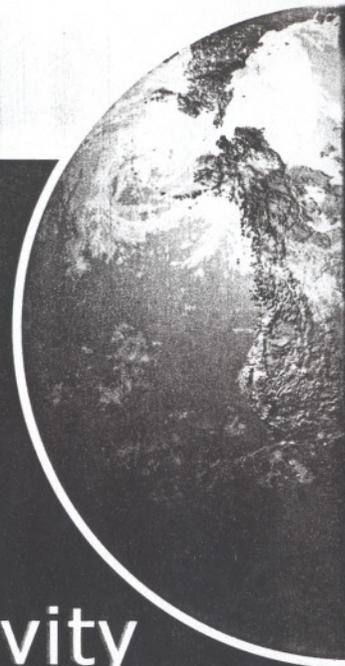
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3 Agricultural Contributions to Greenhouse Gas Emissions

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3.1 Introduction

Agriculture provides both sources and sinks of greenhouse gases (GHGs). The
global intensification of food and fibre production is an important factor
influencing GHG emission. More than 97% of the world's food supply is
produced on land that emits GHGs when intensively tilled and fertilized,
and/or grazed by animals. While US agriculture is generally thought of as a
minor source of GHGs, the increasing world population dictates a challenge to
increase agricultural production without increasing the risks of GHG emissions
and degrading environmental consequences. This review will attempt to put
GHGs from agriculture in perspective, and briefly address fossil fuel in
agriculture, soil carbon (C) loss from intensive tillage, emissions associated
with fertilizers, emissions from animal production and manure management,
and emissions associated with rice production. It has been estimated that 20%
of the greenhouse effect (radiative forcing) is related to agricultural activities
(Cole *et al.*, 1996). Other recent reviews on agriculture's contribution to GHGs
and global change were presented by Houghton *et al.*, 1996; Cole *et al.*, 1997;
Lal, 1997; Lal *et al.*, 1997a,b, 1998; Paul *et al.*, 1997; Paustian *et al.*, 1997a, 1998;
and Rosenzweig and Hillel, 1998. Since the industrial revolution, the inflow
and outflow of carbon dioxide have been disturbed by humans; atmospheric
CO_2 concentrations ($[CO_2]$) have risen about 28% – principally because of
fossil fuel combustion, which accounts for 99% of the total US CO_2 emissions
(Houghton *et al.*, 1996). Agricultural activity, such as clearing forest for fields
and pastures, transforming virgin soil into cultivated land, growing flooded
rice, producing sugarcane, burning crop residues, raising cattle, and utilizing N
fertilizers, are all implicated in the release of GHG into the atmosphere. The
radiative forcing of GHGs and their relative amounts are shown in Fig. 3.1.
Although CO_2, methane (CH_4), and nitrous oxide (N_2O) occur naturally in the
atmosphere, their recent build-up is largely a result of human activities. Since
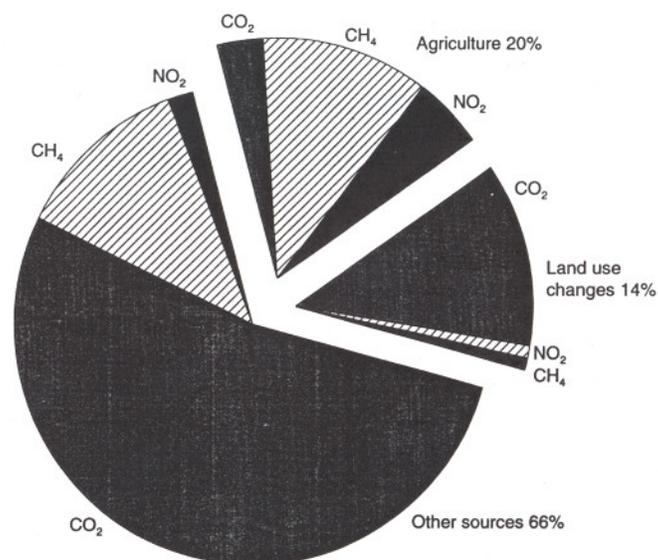
Fig. 3.1. Relative proportions of annual increase in global radiative forcing attributed to agriculture and land use change. (After Cole *et al.*, 1996.)

the 19th century, the atmospheric concentration of these greenhouse gases has increased by 30% for CO_2, 145% for CH_4 and 15% for N_2O (Houghton *et al.*, 1996).

The concept of global warming potential (GWP) has been developed by the Intergovernmental Panel on Climatic Change (IPCC) (Houghton *et al.*, 1996) to compare the ability of GHGs to trap heat in the atmosphere relative to CO_2. The GWP of a greenhouse gas is the ratio of radiative forcing from a unit mass of the gas to a unit mass of CO_2 over a 100-year period. The GWP for $CO_2 = 1$, for $CH_4 = 21$, and for $N_2O = 310$ (see Chapter 2, this volume). Man-made gases such as hydrofluorocarbons, perfluorocarbons and sulphur hexafluoride have significantly higher GWP, but are not from agricultural sources. To quantify the relative amounts of GHGs, IPCC (Houghton *et al.*, 1996) has chosen to express the GHGs in units of million metric tons of carbon equivalents (MMTCE), calculated as the products of the mass of gas (in teragrams Tg) $\times$ GWP $\times$ 12/44. The value of 12/44 is the ratio of the mass of C to the mass of CO_2. For consistency throughout this chapter, GHG units will be expressed as MMTCE. For the amount of CO_2 equivalent, the quantities can be multiplied by 3.67.

The global C cycle is made up of large C reservoirs (or pools) and flows (or fluxes) important to agriculture. The reservoirs of C are interconnected by pathways of exchange through various physical, geological and biological processes. Hundreds of billions of tons of C, in the form of CO_2, are absorbed

by the oceans and by living biomass (through plant photosynthesis), considered to be C sinks. Comparable amounts are emitted to the atmosphere through natural and man-made processes, considered to be C sources. When the system is at equilibrium, the C fluxes or flows among the various pools are roughly balanced. Estimates of these pools are provided in Table 3.1. The largest pool is the oceans, which contain 38 million MMTCE. A large amount of soil C is stored as soil organic matter (SOM) in agricultural production systems. Over most of the earth's land surface, the quantity of C as SOM ranges from 1.4 million to 1.5 million MMTCE and exceeds, by a factor of two or three, the amount of C stored in living vegetation, estimated to be 560,000 MMTCE (Schlesinger, 1990; Eswaran *et al.*, 1993). The contribution of CO_2 released to the atmosphere from agricultural land represents 20–25% of the total amount released due to human activity (Duxbury, *et al.*, 1993). The amount of organic C contained in soils depends on the balance between the inputs of photosynthetically fixed C that go into plant biomass and the loss of C through microbial decomposition. Agricultural practices can modify the organic matter inputs from crop residues and their decomposition, thereby resulting in a net change in the flux of CO_2 to or from soils.

3.2 Fossil Fuel Use in Agriculture

Energy is required for all agricultural operations. Modern intensive agriculture requires much more energy input than did traditional farming methods, since it relies on the use of fossil fuels for tillage, transportation and grain drying, for the manufacture of fertilizers, pesticides and equipment used as agricultural inputs, and for generating electricity used on farms (Frye, 1984). Early estimates suggested that fossil fuel usage by agriculture, primarily of liquid fuels and electricity, constitute only 3–4% of the total consumption in developed countries (CAST, 1992; Enquête Commission, 1995). To provide a reference for agriculture's contribution, C emissions from fossil fuel use in the USA in 1996 were reported to be 286.7, 229.9, 477.5 and 445.5 MMTCE for residential, commercial, industrial and transportation sectors, respectively (EPA, 1998). The total amount of C emitted as CO_2 in the USA in 1996 from

Table 3.1. Estimates of global carbon pools.

Pool	Total C content (MMTCE)[a]	
	Bouwman, 1990	Eswaran *et al.*, 1993
Atmosphere	720,000	750,000
Biomass	560,000–835,000	550,000
Soil organic matter	1,400,000–2,070,000	1,500,000
Caliche[b]	780,000–930,000	–
Oceans	38,000,000	38,000,000

[a]Million metric tonnes of carbon equivalent.
[b]Petrocalcic horizons in arid and semi-arid regions.

fossil fuels was 1450.3 MMTCE, a value that has steadily increased with time. Revised estimates by Lal *et al.* (1998) showed that US agriculture has contributed 116 of the total 1596 MMTCE (i.e. 7.3%) of US emissions. These agricultural emissions include an additional 15 MMTCE due to soil erosion (not included in earlier estimates) and 27.9 MMTCE due to direct on-farm energy use and indirect fertilizer and pesticide production.

Tillage and harvest operations account for the greatest proportion of fuel consumption within intensive agricultural systems (Frye, 1984). Fuel requirements using no-till or reduced tillage systems were 55 and 78%, respectively, of that used for conventional systems that included mouldboard ploughing. On an aerial basis, savings of 23 kg C ha^{-1} per year in energy costs resulted from the conversion of conventional till to no-till. For the 186 Mha of cropland in the USA, this translates into potential C savings of 4.3 MMTCE per year. Kern and Johnson (1993) calculated that conversion of 76% of the cropland planted in the USA to conservation tillage could sequester as much as 286–468 MMTCE over 30 years and concluded that US agriculture could become a net sink for C. Lal (1997) provided a global estimate for C sequestration from conversion of conventional tillage to conservation tillage that was as high as 4900 MMTCE by 2020. Combining economics of fuel cost reductions and environmental benefits of conversion to conservation tillage is a positive first step for agriculture toward decreasing C emissions into the atmosphere.

A summary of the fossil fuels used in US agriculture in 1996 is presented in Table 3.2. The three major fuels used released more than 19 MMTCE directly in 1995, with diesel fuel being the largest contributor. Nitrogen fertilizers, which require the greatest amount of energy to produce, are used in larger amounts than any other fertilizer. Net energy use in fertilizer manufacture has declined up to 40% recently, due to substantial improvements in plant efficiencies and use of natural gas. Estimates of energy required are 45.5, 10.8 and 5.0 Btu g^{-1} of product for N, P$_2$O$_5$ and K$_2$O, respectively (Shapouri *et al.*, 1995). For example, converting from Btu to joules (J) (1 Btu = 1055.06 J) and using the C content for natural gas (13.6 kg C 10^{-9} J) yields 0.66 tons of C released per ton of N produced. Therefore, the annual global consumption of about 80 Tg of fertilizer N corresponds to the consumption of about 53 MMTCE released as CO$_2$. Fertilizer and chemical production in the USA has increased steadily since the 1940s, and contributed 8.3 MMTCE to the atmosphere in 1996. These combined estimates of fossil fuels used in US agriculture represent about 2% of the total US C emissions. However, this table does not include energy for electricity used for heating and cooling or energy for equipment manufacture. Pimentel and Heichel (1991) estimated that the energy required for making agricultural machinery is equal to the fuel used to grow the crop.

Pimentel (1984) indicated that 17% of the total energy used in the US economy is consumed in food systems, with about 6% for agricultural production, 6% for processing and packaging and 5% for distribution and preparation. This 17% of the total US energy use represents an annual per capita use of about 1500 litres of fuel just for food. Taking the C content of fuel oil as 0.73 kg C l^{-1} and multiplying by an estimated US population of 270 million in mid-1998 suggests that food production, processing and

Table 3.2. US agricultural emissions of carbon from use of fossil fuels directly in 1995 and indirectly for chemical inputs in 1996.

Fuels[a]	Volume (10^6 l)	Carbon (MMTCE)
Diesel	13,626	11.02
Gasoline	5,626	3.32
Propane	3,028	4.97
Sum		19.31
Chemicals[b]	Weight (kg × 10^6)	Carbon[c]
Nitrogen (N)	6,916.9	6.30
Phosphorus (P$_2$O$_5$)	2,904.0	0.61
Potash (K$_2$O)	3,181.0	0.31
Herbicide	155.1	0.68
Insecticide	18.8	0.08
Fungicide	3.32	0.01
Other chemicals	59.1	0.26
Sum		8.25

[a]Source: Agricultural Resources and Environmental Indicators #16, Dec. 1996; USDA-ERS, Office of Energy, based on data gathered by NASS.
[b] Source: USDA-NASS, Agricultural Statistics Board. 1996 Field Crops Summary, Agricultural Chemical Usage.
[c]Assuming fuel was natural gas at 14 mg C (Btu)$^{-1}$. Represents 100% of land farmed (123,968,000 ha), which was extrapolated from a survey that covered seven major crops and 71.5% of land.

preparation would emit about 296 MMTCE per year in the USA, or about 20% of the US total (Houghton *et al.*, 1996). Fossil fuel requirements by the food sector as a whole (which includes processing, preservation, storage and distribution) account for 10–20% of the total fossil energy consumption (Pimentel *et al.*, 1990; CAST, 1992). Thus, mitigation of energy use by agriculture should consider the 11% in non-production areas when considering solutions for decreasing the amount of CO$_2$ emitted by agriculture.

3.3 Management of Soil Carbon

Conversion of forest land to agricultural land or urban use can result in changes in emissions of soil C as CO$_2$. Conversely, net additions of forest and crop biomass can result in soil acting as a sink for CO$_2$ (Raich and Potter, 1995). Agriculture and intensive tillage have caused a decrease of between 30 and 50% in soil C since many soils were brought into cultivation more than 100 years ago (Schlesinger, 1986; Houghton, 1995). There needs to be a better understanding of tillage processes, the mechanisms leading to C loss and how this C loss can be linked to soil productivity, soil quality, C sequestration, and ultimately to crop production. Long-term studies of soil C point to the role of

intensive tillage and residue management in soil C losses (Lal, 1997; Paul *et al.*, 1997; Paustian *et al.*, 1997b); however, extrapolation of these data to a global value is complicated by uncertainties in soil C quantities and distribution across the landscape. Paustian *et al.* (1998) estimated that better global management of agricultural soils, restoring degraded soils, permanent set-aside of surplus land and restoration of some wetlands now used for agriculture could sequester between 400 and 900 MMTCE per year in the soil. They caution that soils have a finite capacity to store additional C which likely will be realized within 50–100 years. The potential for improved management offers hope that agriculture can decrease GHG emissions.

Mineral soils generally have fairly shallow organic layers and, therefore, have low organic C content relative to organic soils (Lal *et al.*, 1997a; Paustian *et al.*, 1997b). Consequently, it is possible to deplete the C stock of a mineral soil within the first 10–20 years of tillage, depending on type of disturbance, climate and soil type. Once the majority of native C stocks have been depleted, an equilibrium is reached that reflects a balance between accumulation from plant residues and loss of C through decomposition. Lal (1997) calculates that if 15% of the C in crop residues is converted to passive soil organic C (SOC), it may lead to C sequestration at the rate of 200 MMTCE year[-1] when used with less intensive tillage. If the current changes in improved residue management and conversion from conventional tillage to conservation tillage in mineral soils continue as they have in the recent decade, these changes may lead to cumulative global C sequestration that ranges from 1500 to 4900 MMTCE by the year 2020 (Lal, 1997). In addition to increasing SOM, combined ecological and economic benefits of conservation tillage also accrue from decreased soil erosion, lower energy costs, water conservation and quality improvements, soil temperature regulation and improved soil structure. These all contribute to enhanced environmental quality and increased crop production.

One example of what intensive tillage in agricultural production systems has done to soil organic C is illustrated in Fig. 3.2. These data illustrate the long-term trends in soil C at the Morrow plots in Champaign, Illinois (Peck, 1989), and Sanborn Field at the University of Missouri, Columbia, Missouri. (Wagner, 1989). Both locations show similar decreases in SOC over the last 100 years. The only experimental parameter or factor common to the two locations was use of a mouldboard plough to till the experimental plots. Different cropping systems or rotations yielded a difference in soil C, which shows that management options exist for controlling SOM and improving soil C levels. The large decline in soil C was a result of tillage-induced soil C losses caused by use of the mouldboard plough and disk harrow, and a change to annual species. Other work around the world shows similar trends (Lal, 1997; Paul *et al.*, 1997; Paustian *et al.*, 1997b) and supports the need for conservation tillage with improved residue management. The significant 'flush' of CO_2 immediately after tillage reported by Reicosky and Lindstrom (1993, 1995) partially explains the long-term role of tillage in affecting C flow within agricultural production systems. Tillage, particularly mouldboard ploughing, resulted in a loss of CO_2 within minutes of tillage. Nineteen days after mouldboard ploughing, C lost as CO_2 accounted for 134% of the C in the
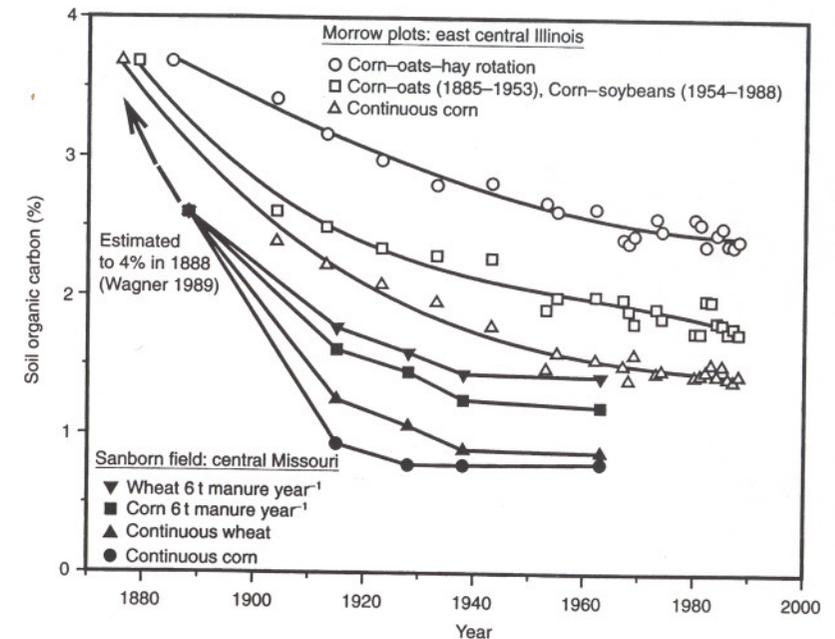
Fig. 3.2. Long-term effects of tillage and crop rotations on soil carbon in Midwest USA.

previous wheat residue. Mouldboard ploughing, one of the most disruptive types of tillage, appears to have two major effects: (i) to loosen and invert the soil, allowing rapid CO_2 loss and O_2 entry into the soil; and (ii) to incorporate/mix the crop residues, thus enhancing microbial attack. Tillage perturbs the soil system and causes a shift in the gaseous equilibrium by releasing CO_2 that enhances oxidation of soil C and organic matter loss. Conservation tillage, or any form of less intensive tillage, can minimize this tillage-induced C loss (Lal, 1997; Paustian *et al.*, 1997b).

Sustainable agriculture requires new technologies for efficient biomass C utilization. Crop stover or residue is an important and renewable resource that is manageable and serves as the primary input for soil C sequestration. Lal (1997) has estimated that the global arable land mass of about 1.4×10^9 ha annually produces 3.44×10^9 Mg of crop residue. At mean C content of 45 g kg^{-1} residue, the total global C assimilation is about 1500 MMTCE year[-1]. While a large portion of crop residue C is recycled to CO_2 through microbial decomposition when the residue is mixed with soil by tillage, a small portion remains as humus that contributes to long-term sequestration in soil. The C from agricultural crop residues is only a small fraction (1%) of the estimated total global C fixed in photosynthesis; however, it is one amenable to management.

3.4 Nitrous Oxide and Methane Emissions from Animal Wastes and Lagoons

Nitrous oxide is produced from a wide variety of biological sources in soil, water, and animal wastes. During the last two centuries, human activities have increased N_2O concentration by 13% (EPA, 1998). The main activities producing N_2O are fossil fuel combustion, agricultural soil management and industrial sources. Use of large amounts of N fertilizer creates secondary problems associated with N_2O released in anaerobic conditions (Mosier *et al.*, 1998a). Agricultural soil management activities such as fertilizer application and cropping practices were the largest source of N_2O emission (56.5 MMTCE), accounting for 43% of the US total (EPA, 1998). Manure management in feedlots (3.7 MMTCE) and agricultural residue burning (0.1 MMTCE) are small sources of N_2O emissions.

Methane is second only to CO_2 in contributing to GHG emissions. Landfills are the largest contributor to CH_4 emissions in the USA, while the agricultural sector is responsible for 30% of US emissions. Of the total 176.7 MMTCE emitted in the USA in 1996 (EPA, 1998), agricultural emissions of CH_4 were: ruminant livestock fermentation, 34.5 MMTCE; agricultural manure management, 16.6 MMTCE; rice cultivation, 2.5 MMTCE; and biomass burning (Mosier *et al.*, 1998b).

Greenhouse gases are associated with storage and application of animal manure. Of these GHGs, the greatest attention has been given to CH_4 emissions generated by animals. There has been very little attention given to CO_2 production by manure storage systems. Among the agricultural sector's potential CH_4 emission sources, manure appears to contribute approximately 5% of the total (Table 3.3). Within the manure portion of CH_4 emissions, swine production constitutes the largest amount due to the type of manure handling and storage (Table 3.4). Nitrous oxide generation within manure is a result of the nitrification/denitrification process that occurs in manure storage and application. After field application, it would be difficult to separate the N_2O from manure sources from that of commercial fertilizer sources in the soil. Methane has been the gas most often measured in various studies; however, data comparing different production practices are sparse.

Table 3.3. Agricultural sources of atmospheric methane emissions in the US (EPA, 1994a).

Source	Methane emission	
	(Tg year^{-1})	(MMTCE)
Rice	65	372.3
Livestock	80	458.2
Manure	10	57.3
Biomass burning	30	171.8
Sum	185	1059.6

Table 3.4. Methane emissions from livestock manure in the USA and the World (EPA, 1994a,b).

Species	Methane emissions			
	USA (Tg year^{-1})	World (Tg year^{-1})	USA (MMTCE)	World (MMTCE)
Dairy	0.71	2.89	4.07	16.55
Beef	0.19	3.16	1.09	18.09
Swine	1.11	5.29	6.36	30.16
Sheep and goat	–	0.71	–	4.07
Poultry	0.23	1.28	1.32	7.33
Other	0.24	0.51	1.37	2.92
Sum	2.48	13.84	14.21	79.12

Production of GHG from manure storage systems has not been sufficiently measured over a large number of units and over a wide range of climatic conditions. Methane production within lagoons and earthen storage systems comes from the solid/liquid interface, with the CH_4-producing bacteria present at this interface. Anaerobic digestion of manure leads to the production of CH_4. Hill and Bolte (1989) described the anaerobic manure storage system as a complex set of interdependent biological systems. Methane production is part of the biological complex, and they proposed that loading rate, pH and temperature were factors causing shifts in the balance among the organisms. An illustration of these interactions is given by Burton (1992), who found that shifting the anaerobic manure storage to an aerobic storage reduced the potential NH_3 loss to the atmosphere. Unfortunately, this shift can lead to production of N_2O. However, he did not quantify the expected release of these gases. Safley *et al.* (1992) characterized the emission of CH_4 from different livestock systems and concluded that anaerobic manure storage systems would convert non-lignin organic matter into CH_4 under warm, moist, anaerobic conditions. Parsons and Williams (1987) developed a mathematical model for anaerobic storage systems based on these factors that could be adapted for prediction of GHG.

The annual release of CH_4 from different manure storage systems associated with swine vary from 10 kg per animal for subconfinement pits within buildings to about 90 kg per animal in a lagoon system. This variation in CH_4 production can be attributed to the amount of solids in the different manure systems and the bacterial populations present in the manure storage. Groenestein and Faassen (1996) found that deep-litter systems for swine reduced N_2O emissions because of changes in the manure digestion systems within the manure. Changes in manure management have had a positive impact on emission rates. Prueger and Hatfield (unpublished data, 1997) positioned a trace gas analyser over a lagoon and found that there was variation in the CH_4 fluxes throughout the day in response to diurnal changes in temperature. (Similar variation has been observed in rice fields; Sass *et al.*, 1991b; Satpathy *et al.*, 1997; Wang *et al.*, 1997a.) Prueger and Hatfield also

documented variation in the exchange coefficient between the lagoon surface and the atmosphere. However, these data were not collected for a sufficient length of time to quantify seasonal changes in CH_4 production in response to a wide range of atmospheric conditions. Kinsman *et al.* (1995) measured CH_4 and CO_2 production from lactating dairy cows and found that stored manure contributed 5.8 and 6.1%, respectively, to CH_4 and CO_2 emissions under conditions of their experiment. Manure storage, particularly for ruminant-animals, represents a small fraction of the total GHG load to the atmosphere.

Kuroda *et al.* (1996) measured the emissions of GHG emitted during composting of swine faeces under continuous aeration using laboratory-scale composting apparatus. Methane emission was observed within only 1 day from starting the composting, while N_2O and NH_3 repeatedly rose and fell after every turning. Of the total N loss during composting, the total amount of N_2O emission was a small fraction of NH_3 emissions. Lessard *et al.* (1996) measured N_2O emissions from agricultural soils after application of dairy cattle manure to cultivated land planted to maize (*Zea mays* L.). The manure application rates were 0, 170 and 339 kg N ha^{-1}, respectively. On the manured plots, 67% of the total N_2O emitted during the growing season occurred during the first 7 weeks following manure application. High N_2O fluxes coincided with periods when NO_3-N levels and soil water contents were relatively high. Fluxes were highest the first day after manure application, but returned to near pre-application levels 7 days later. There were short-lived peaks of N_2O flux, usually following rain. Only 1% of the manure N, which accumulated as N_2O, was potentially mineralizable over the snow-free season. In a similar study, Wassman *et al.* (1996) evaluated the effect of fertilizers and manure on CH_4 emission rates using an automated, closed-chamber system in Chinese rice (*Oryza sativa*) fields. The rate of increase in CH_4 emission was dependent on the total amount of organic manure applied. A single application of organic manure increased the relative short-term CH_4 emission rates by 2.7–4.1 times compared with fields without organic manure.

Reports on the literature indicate that there is a large amount of variation in the fluxes of GHG from animal manure storage and handling. These differences could be attributed to variations in species, diet, loading rates into the storage, type of storage and environmental conditions within the manure storage. Further studies giving greater attention to the physical and biological parameters affecting microbial production and emission of GHG are needed. These data will have to be coupled with dietary models for different species and a complete understanding of the chemical factors within manure storage systems in order to quantify the dynamics of GHG production and emission. This type of information will be essential in developing realistic mitigation scenarios.

3.5 Rice and Methane Production

Agricultural sources of CH_4 account for as much as one-third of the total atmospheric pool, with a significant portion contributed by rice cultivation.

A recent estimate suggests that CH_4 emitted from global rice paddies is 60 ± 40 Tg year^{-1} (344 ± 229 MMTCE) (Houghton *et al.*, 1992). Methane emission from rice fields is the result of bacterial processes – production in flooded anaerobic microsites and consumption (oxidation) in aerobic microsites). Flooding of rice fields promotes anaerobic fermentation of C sources supplied by the rice plants and other incorporated organics, resulting in the formation of CH_4. The process is governed by a complex set of parameters linking the physical and biological characteristics of flooded soil environments with specific agricultural management practices.

Rice is grown under a variety of climatic, soil and hydrological conditions in nearly 90 countries, and rice production can conveniently be divided into four categories based on water availability and CH_4 emission. The relative source strengths of CH_4 from these four rice production systems are: irrigated rice and favourable rainfed rice > flood-prone rainfed rice and deep-water rice > drought-prone rainfed rice > tidal wetland rice. Upland rice is not a source of CH_4 since it is grown on aerated soils (Neüe, 1997). Several other reviews on CH_4 emissions from rice fields have been published (Cicerone and Oremland, 1988; Neüe, 1993, 1997; Neüe and Sass, 1994; Sass and Fisher, 1996).

Conditions for CH_4 production in wetland rice soils have been categorized into six areas: water regime; Eh (redox potential)/pH buffer; carbon supply; temperature; texture and mineralogy; and salinity (Neüe, 1997). Methane production was influenced both by the reduction characteristics of the soils and by labile organic substrates (Gaunt *et al.*, 1997) and texture (Parashar *et al.*, 1991; Chen *et al.*, 1993). Sass *et al.* (1994) found a strong linear correlation between seasonal CH_4 emission and the percentage of sand in a sand : clay : silt gradient among three soils in Texas. In general, sandy soils high in organic C produce more CH_4 than clay soils with similar or lower C content (Neüe and Sass, 1994). Significantly decreased CH_4 emissions have been observed in soils with high percolation rates (Inubushi *et al.*, 1992). Increased percolation may transport sufficient dissolved oxygen in the soil to raise the Eh sufficiently either to inhibit CH_4 production or to increase CH_4 oxidation.

Variations in seasonal CH_4 emission from rice paddies are complex. A correlation with soil temperature has been reported in some studies, but not in others (Wang *et al.*, 1990; Neüe *et al.*, 1994; Neüe and Sass, 1994). Seasonal CH_4 fluxes observed in temperate rice fields show a general seasonal trend related to plant development (Sass *et al.*, 1991a,b, 1992). Methane emissions show a gradual rise during the vegetative phase that correlates with increasing plant biomass and peaks near panicle differentiation. This peak in emission may be attributed to a stabilization of soil pH and redox potential, root porosity, and an increasing amount of C substrate (Neüe and Sass, 1994; Kludze *et al.*, 1993). Prior to the end of the season, a second emission peak is sometimes observed which may be attributed to an increase in soil C due to leaf and root senescence (Neüe and Sass, 1994).

In irrigated double-cropped (two crops in one year) tropical rice paddies, both CH_4 emission and grain yield are consistently higher from the dry season

crop than from the wet season crop (Neüe *et al.*, 1994). These results suggest that higher photosynthetic rates during the sunnier days of the dry season lead to larger amounts of C available to methanogenic bacteria and, consequently, to greater CH_4 emission rates. The addition of readily degradable C in sources such as rice straw before planting results in an additional early-season peak in CH_4 emission as the straw rapidly decomposes (Lindau *et al.*, 1991; Neüe and Sass, 1994). Other forms of C added by farmers, either for fertilization or to dispose of non-grain biomass, tend to increase both CH_4 production and emission (Sass *et al.*, 1991a,b; Neüe *et al.*, 1994; Minami, 1995). The incorporation of green manure leads to even higher emission levels (Denier van der Gon and Neüe, 1995).

Fertilizer is necessary to ensure adequate rice growth and root development. Wassmann *et al.* (1996) investigated the effect of fertilizers on CH_4 emission rates in Chinese rice fields, and found the rate of increase in CH_4 emission depended on the amount and timing of organic manure application. A potential mitigation technique involving double cropping was observed by these authors. Organic amendments are applied to the first rice crop (low CH_4 emission rates), and exclusively mineral fertilizers are applied to the second crop (high CH_4 emission rates). This fertilization distribution pattern does not reduce yields and results in a combined annual CH_4 emission that is only 56% of that emitted from fields treated with only blended mineral fertilizers over both seasons. Lindau *et al.* (1991) measured increased CH_4 emission with increased urea application in flooded rice fields of Louisiana, USA, where application of 200 kg urea-N ha^{-1} is typical. Similar emissions were measured with applications of either 200 or 300 kg urea-N ha^{-1}. However, emissions were lower when less than 200 kg urea-N fertilizer ha^{-1} was used. A reduction in CH_4 emission when ammonium sulphate fertilizer was used may be due to substrate competition by sulphate-reducing bacteria or to hydrogen sulphide toxicity (Neüe and Sass, 1994).

Methane emission rates are highly sensitive to water management. Periodic drainage of irrigated rice paddies, a common management practice in Japan, results in a significant decrease in CH_4 emissions (Yagi *et al.*, 1996; Cai *et al.*, 1997). Intermittent irrigation reduced CH_4 emissions by 36% compared with that from constant submergence of soil (Shin *et al.*, 1996). In the Philippines, draining for a period of 2 weeks at mid-tillering stage or at panicle initiation successfully suppressed CH_4 flux by up to 60%. However, N_2O flux increased sharply during the drainage period (Bronson *et al.*, 1997). Sass *et al.* (1992) found that a single mid-season drain reduced seasonal emission rates of CH_4 by 50%, and multiple short periods of drainage (2–3 days) reduced CH_4 emissions to an insignificant amount.

As shown in China (Yue *et al.*, 1997), an important contributor to variation in measured CH_4 emissions may be the use of different rice cultivars. Semi-dwarf varieties emit significantly less CH_4 than do tall varieties (Lindau *et al.*, 1995). In the Philippines (Neüe *et al.*, 1996; Wang *et al.*, 1997b), CH_4 emission rates from different cultivars showed a high correlation with root dry weight and total C released from roots. Cultivar-dependent variation in seasonal CH_4 emissions ranged from 18 to 41 g m^{-2} (Sass and Fisher, 1996;

Huang *et al.*, 1997a). Emission from a newly developed high-yielding, low tillering cultivar (IR65598) was very low. These differences in CH_4 emissions are attributed to differences in gas transport capacity among cultivars (Butterbachbahl *et al.*, 1997). Farmers' choice of the appropriate rice cultivar can therefore influence regional and global emissions of CH_4 without adversely affecting grain yields.

Because rice is an important crop globally, GHG mitigation efforts suggested by Yagi *et al.* (1997) must be based on sound agricultural practices and good science. Estimates of CH_4 emissions have been made in the following ways: by extrapolating field measurements to a regional or global scale (Wang *et al.*, 1994) assuming CH_4 emission as a constant fraction of rice net primary productivity (Bachelet and Neüe, 1993; Bachelet *et al.*, 1995); or by correlating CH_4 emissions with production (Anastasi *et al.*, 1992) or with organic matter inputs (Kern *et al.*, 1995). A field trial suggested that CH_4 emission can be predicted by a model utilizing environmental variables particular to a given region (Cao *et al.*, 1995). A semi-empirical model which predicts daily CH_4 emission from flooded rice fields (Huang *et al.*, 1998) is based on studies in Texas (Sass *et al.*, 1991a,b, 1992, 1994; Sass and Fisher, 1995; Huang *et al.*, 1997a,b; Sigren *et al.*, 1997a,b). Future research for mitigation will be directed toward using models, along with ground-truth, to interpret satellite-based sensor data for accurate assessments of regional, national and global trace-gas emissions from rice agriculture.

3.6 Mitigation Options for Agriculture

What agriculture can do to mitigate GHG emissions has been estimated by Cole *et al.* (1997), whose estimates of potential reduction of radiative forcing by the agricultural sector range from 1150 to 3300 MMTCE year^{-1}. Of the total potential global reduction in GHG emissions, approximately 32% could result from reduction in CO_2 emissions, 42% of the C offsets from biofuel production on 15% of the existing croplands, 16% from reduced CH_4 emissions and 10% from reduced emissions of N_2O.

Agriculture can contribute to mitigation of climatic change by adopting practices that promote stashing CO_2 as C in soil, crop biomass and trees, and by displacing the use of fossil fuels required for tillage, chemical manufacture equipment manufacture, and grain handling operations (Cole *et al.*, 1996; Paustian *et al.*, 1998). For the farm sector, the GHG mitigation potential through reduced fuel consumption is relatively small when compared with the rest of society; however, further reductions can be achieved. By combining appropriate land with best management practices to increase US crop production, Lal *et al.* (1998) suggest a soil C sequestration potential of 126 MMTCE year^{-1}. Much of the potential C sequestration (43% of US potential) comes from conservation tillage and crop residue management. Other strategies include eliminating fallow by using cover crops, improved irrigation scheduling, solar drying of crops, improved soil fertility, improved manure management, and producing more food using less land. Optimizing

N fertilizer efficiency, achieving higher yield per unit land area and using conservation tillage hold the most promise for indirectly mitigating N_2O and CO_2 emissions. Mitigation of CH_4 emissions from agriculture will require improved diets and rations for animals, aerobic conditions in manure management and improved rice production. Practices that will have the most impact on GHGs from rice production are water and carbon management, soil and variety selection, fertilizer type and amount, and soil preparation. Global understanding of these critical management practices will lead to enhanced soil and plant management and the development of new technologies that result in increased food production efficiency with minimum impact on environmental quality and GHGs. Acceptance of mitigation options will depend on the extent to which sustainable agricultural production can be achieved and the combined social, economic and environmental benefits.

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