Review

Agricultural opportunities to mitigate greenhouse gas emissions[☆,☆☆]

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Management options can be used to reduce agriculture's environmental impacts.

Abstract

Agriculture is a source for three primary greenhouse gases (GHGs): CO_2, CH_4, and N_2O. It can also be a sink for CO_2 through C sequestration into biomass products and soil organic matter. We summarized the literature on GHG emissions and C sequestration, providing a perspective on how agriculture can reduce its GHG burden and how it can help to mitigate GHG emissions through conservation measures. Impacts of agricultural practices and systems on GHG emission are reviewed and potential trade-offs among potential mitigation options are discussed. Conservation practices that help prevent soil erosion, may also sequester soil C and enhance CH_4 consumption. Managing N to match crop needs can reduce N_2O emission and avoid adverse impacts on water quality. Manipulating animal diet and manure management can reduce CH_4 and N_2O emission from animal agriculture. All segments of agriculture have management options that can reduce agriculture's environmental footprint.
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1. Introduction

Human activities including modern agriculture contribute to the production of greenhouse gases (GHGs), which have increased since the advent of the industrial age (IPCC, 1996). Greenhouse gases are defined by their radiative forcing, which changes the Earth's atmospheric energy balance; typically, expressed as watts per square meter ($W\,m^{-2}$) (IPCC, 1996). A positive value indicates an increase in the level of energy remaining on the Earth, while a negative value indicates an increase in the level of energy returning to space. Climate forcing potential or global warming potential (GWP) is a function of radiative forcing (i.e., the expected effect from the addition of a unit of gas on the radiation balance of the Earth), mean lifetime (i.e., how long the forcing by a unit of gas is expected to continue) and emissions (i.e., total quantity of gas emitted). The first two factors can be defined as GWP. Several naturally produced GHGs trap heat, including water vapor, carbon dioxide (CO_2), ozone (O_3), methane (CH_4) and nitrous oxide (N_2O). Other GHGs (e.g., hydrofluorocarbon, perfluorocarbons and sulfur hexafluorides) are solely the result of human activity (IPCC, 1996). Carbon dioxide, CH_4, and N_2O are long-lived in the atmosphere and are the major contributors to positive increases in radiative forces (IPCC, 1996). Agricultural activities are significant producers of CH_4 and N_2O. Of the three main gases that are influenced by land management and that are responsible for the potential greenhouse effect,

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CO_2 has the greatest climate forcing potential (57%), while CH_4 and N_2O account for 27% and 16%, respectively (CAST, 1992). This review focuses on CO_2, CH_4 and N_2O emission especially in North America as related to agriculture.

1.1. History of global climate change research

Impacts of anthropogenic activity on increased atmospheric CO_2 concentration and global climate have been discussed for over 100 years. In 1861 John Tyndall stated that CO_2 could effectively trap heat (Tyndall, 1861; Weart, 2004). Other early work led to a better understanding of the relationship between atmospheric CO_2 concentration and global temperature (Arrhenius, 1896; Bolin and Eriksson, 1959; Callendar, 1938; Plass, 1959; Revelle and Suess, 1957). Keeling (1960, 1976) provided accurate background data on atmospheric CO_2 concentration, thereby improving the ability to subsequently document increases in CO_2 concentration. Ice core data of historical atmospheric CO_2 concentration demonstrated a relationship between CO_2 and global temperature (Barnola et al., 1987; Raynaud and Barnola, 1985; Sundquist, 1987). In the 1970s, the GWP of other trace gases (e.g., N_2O, CH_4 and chlorofluorocarbons) became recognized (Lovelock, 1974; Ramanathan, 1975; Wang, 1976). By the 1980s, the anthropogenic influence on global warming gained enough credibility to spark international political activity leading to the establishment of the Intergovernmental Panel on Climate Change (IPCC) (IPCC, 1990) and a plethora of subsequent research. The IPCC (2001, 2007) recently confirmed the anthropogenic influence of GHGs on global climate change.

Pre-industrial levels of atmospheric CO_2 concentration were estimated as 290–295 ppm (Bolin et al., 1979). By 1990, CO_2 concentration had risen to 350 ppm (Wood, 1990), surpassing 370 ppm at the Scripps Institution of Oceanography monitoring sites in 2004 (Keeling and Whorf, 2005). It is predicted that the CO_2 concentration could reach 500 ppm by the end of the 21st century (IPCC, 1996). The IPCC reported that since pre-industrial times the atmospheric concentrations of CH_4 rose 145% and N_2O rose 15% by 1992 (IPCC, 1996). After an apparent reduction of CH_4 flux during the 1990s, anthropogenic emission of CH_4 has increased again (Bousquet et al., 2006). N_2O concentration increased linearly between 1979 and 2004 (Hofmann, 2005).

The dynamics of climate interactions are not completely understood, although there is a general scientific consensus that anthropogenic actions are contributing to global climate change (IPCC, 2001, 2007; Oreskes, 2004). Anticipated and observed impacts of global climate change include increased sea level (Gregory et al., 2001; Shepherd and Wingham, 2007), changes in rainfall distribution and increased storm intensity (IPCC, 2007; Lowe et al., 2001) and accelerated species extinction rate (Thomas et al., 2004). Current strategies for coping with global warming can be divided into two general categories: (1) reducing fossil fuel combustion, as well as curbing emissions of other GHGs and (2) increasing C sequestration (Kimble et al., 2001).

1.2. Major contributors to greenhouse gas emission

Total GHG emission for the United States in 2004 was estimated as 84.6% from CO_2, 7.9% from CH_4, 5.5 from N_2O and 2% from hydrochlorofluorocarbons, chlorofluorocarbons and sulfur hexafluorides (US EPA, 2005). A mole of CO_2 is defined to have a GWP of one; the GWP of other GHGs is higher (Table 1). Therefore, even though non-CO_2 GHGs represent only a small percentage of the GHG mixture, they can make a sizable contribution to the total GWP.

A recent inventory of the United States GHG emission separated major emitters into categories: (1) energy (86.6%), (2) agriculture (6.3%), (3) industrial (4.5%), (4) waste (2.7%) and (5) solvent and product use (<1%) (Fig. 1; US EPA, 2007). Land-use changes and forestry were treated as a sink, offsetting emission by about 12%. Emission of CO_2 from on-farm energy use was included in the energy category (US EPA, 2005, 2007). Agricultural related activities contribute about one-third of the global net CO_2 emissions mostly from deforestation and burning; however, drainage of wetland and tillage also contribute (Flach et al., 1997). There are differences noted among emission sources, caused by inconsistency in categorical definitions.

Within the agricultural sector, CO_2, CH_4 and N_2O are the gases considered to be of primary concern (Reicosky et al., 2000; US EPA, 2005). The greatest contributions to CH_4 emission are enteric fermentation (21%) and manure management (8%) with minor contributions from rice (*Oryza sativa* L.) paddies and agricultural burning (Fig. 2; US EPA, 2007). Application of nitrogenous fertilizer and cropping practices are estimated to cause 78% of the total N_2O emission in the United States (Fig. 3; US EPA, 2007). In the European Union, it was estimated that about 10% of the GHG emission is due to CH_4 and N_2O of which 49 and 63%, respectively, have been attributed to agriculture (Weiske and Petersen, 2006). These estimates can be improved as the magnitude of gas flux from the agricultural sector still has large knowledge gaps (Franzluebbers and Follett, 2005). None-the-less these are sizable contributions, which can be modified through management.

Table 1

Global warming potential over a 100-year time span of several greenhouse gases, CO_2, CH_4, and N_2O are naturally occurring and related to agriculture; hydrochlorofluorocarbons (HFC), chlorofluorocarbons (CFC) and sulfur hexafluorides (SF) are strictly anthropogenic and are not typically associated with an agricultural source

Source	Gas	SAR[a]	TAR[b]
Naturally occurring and agricultural related	CO_2	1	1
	CH_4	21	23
	N_2O	310	296
Strictly anthropogenic – nonagricultural sources	HFC	140–11,700	120–12,000
	CFC	6500–9200	5700–11,900
	SF	23,900	22,200

Specific HFCs and CFCs have different individual GWP values.
[a] IPCC Second Assessment Report (IPCC, 2001).
[b] IPCC Third Assessment Report (IPCC, 2007).

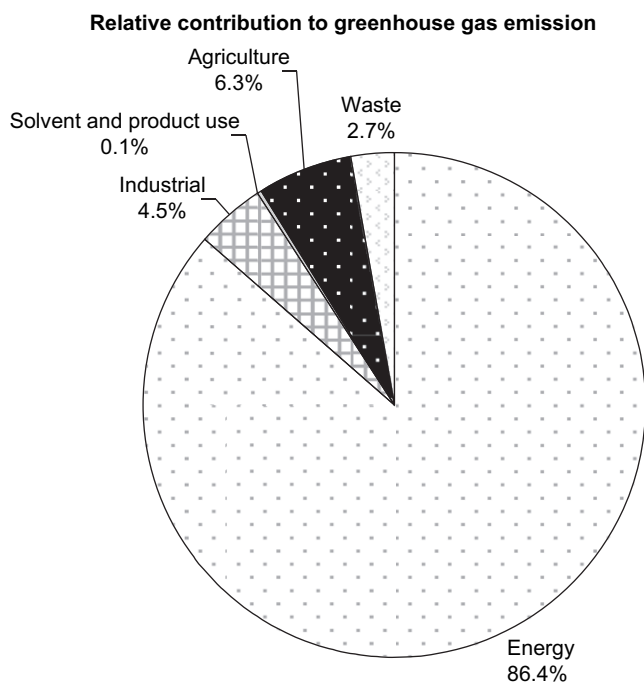**Relative contribution to greenhouse gas emission**

Fig. 1. Relative contribution to greenhouse gas emission by major emitters, in the United States. (US EPA, 2007).

2. Carbon dioxide

Historically, land-use change from native prairie and forest to intensive agriculture has been a contributor to anthropogenically induced CO_2 emission. Prior to the1960s, the release of CO_2 from soil and biota was greater than the release from
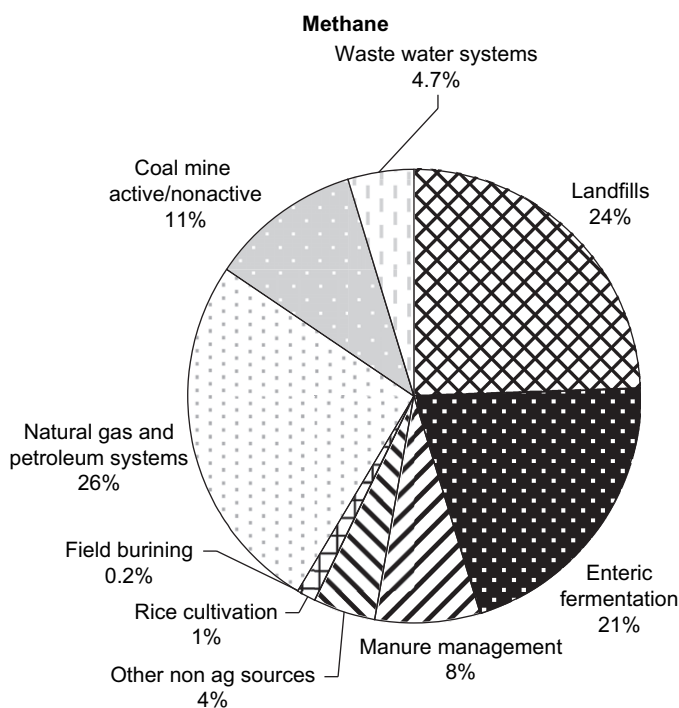
Methane

Fig. 2. Relative contribution to CH_4 emission in the United States (US EPA, 2007).
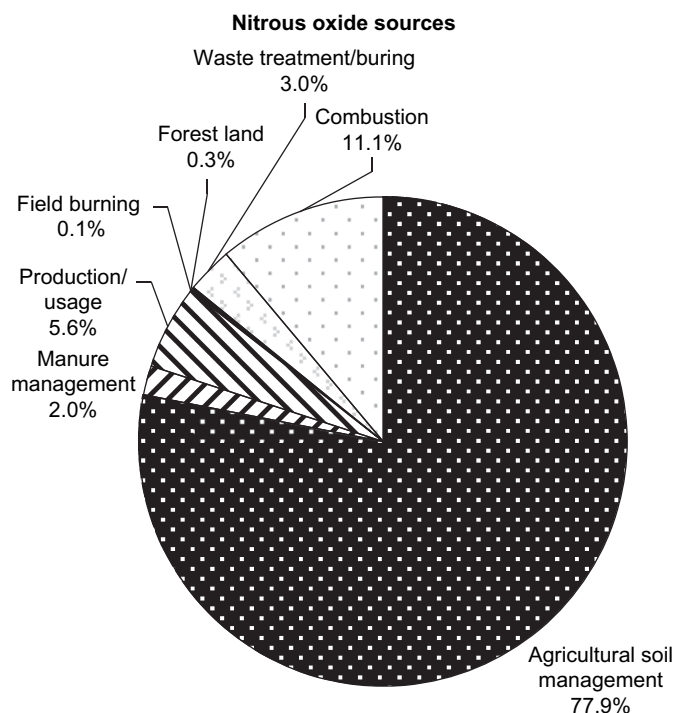
Nitrous oxide sources

Fig. 3. Relative contribution to N_2O emission in the United States (US EPA, 2007).

fossil fuels (Houghton et al., 1983). Agriculture consumes fossil fuels during the manufacture of equipment, fertilizer and other chemical inputs as well as during machinery and grain handling operations (e.g., grain dying). Strides have been made to change agriculture from a net contributor to GHG emission to a net sink via C sequestration (Cole et al., 1997; Lal, 2007; Reicosky et al., 2000).

2.1. Carbon sequestration defined

The Ecological Society of America refers to C sequestration as long-term storage (ESA, 2000). The US-DOE Office of Science (http://cdiac2.esd.ornl.gov/index.html) adds the caveat that C sequestration results in a slowing or reduction of CO_2 (the principal GHG) concentration in the atmosphere (US-DOE, 2002). However, the term "sequestration" is not always used to imply long-term removal of CO_2 from the atmosphere. Colloquial use of "sequestration" equates C capture during photosynthesis as sequestration (Wikipedia Contributors, 2006) without regard to how long the CO_2 is removed from the atmosphere. Much of the photosynthetically fixed CO_2 is returned to the atmosphere during respiration and decomposition. In this review, we define terrestrial C sequestration as net transfer of atmospheric CO_2 into long-lived C pools. These pools can be extant biomass (e.g., trees), long-term biomass products (e.g., lumber), living biomass in soils (e.g., perennial roots and microorganisms), or recalcitrant organic and inorganic C in soils and deeper subsurface environments (West and Post, 2002).

Sequestration of C from plant biomass into soil organic matter (SOM) is a key sequestration pathway in agriculture.

Mean residence time of "stable" or "long-lived" SOM varies from 250 to 1900 years (Stevenson, 1994). For C to be sequestered in the soil it needs to be protected from microbial degradation within stable microaggregates (<250 μm), adsorbed on the inner surfaces of clays or be chemically protected in organo-mineral complexes (Lal, 1997). Mean residence time of SOM is affected by the type of clay, in which 1:1 clays like kaolinite have a shorter turnover time than 2:1 clays like smectite (Wattel-Koekkoek et al., 2003). Mean residence time also tends to increase with depth in the soil profile (Paul et al., 2001). Soil C can also be sequestered in charcoal, which may persist to 1500 years (Lehmann et al., 2006).

Within a given soil and climatic regime, a linear relationship exists between biomass C inputs and Soil organic matter (SOC) (Huggins et al., 1998; Larson et al., 1972; Rasmussen et al., 1980). Maximum C sequestration within a given ecosystem occurs once the inputs and the outputs reach steady-state or dynamic equilibrium (Lal et al., 1998). Ingram and Fernandes (2001) noted that apparent differences among management systems are a function of time, suggesting terminology such as attainable$_{max}$ C level when addressing management effects on soil C levels.

2.2. Benefits of sequestering C

There are multiple benefits to sequestering C in forest and agricultural soils, beyond the obvious benefit of offsetting CO_2 emissions. Lal (2007) summarized collateral soil C sequestration benefits on improved soil quality, increased soil productivity, reduced risk of soil erosion and sedimentation, decreased eutrophication and water contamination. Soil organic matter is about 58% C with a C:N ratio between 10 and 12 (Stevenson, 1994). Increasing SOM increases both C and N concentration in the soil. Many physical, chemical, and biological characteristics associated with productive soil are due to high SOM content (Doran, 2002; Doran et al., 1998; Janzen et al., 1998). Soil aggregation and aggregate stability are improved by SOM (Gollany et al., 1991; Pikul et al., 2005; Six et al., 1998; Tisdall, 1996; Tisdall and Oades, 1982). Increasing SOM also improves water infiltration, water-holding capacity, aeration, bulk density (Gollany et al., 1992; Olness and Archer, 2005), penetration resistance and soil tilth. Soil organic matter plays an important role in determining soil chemical properties including pH, nutrient availability and cycling, cation exchange capacity and buffer capacity (Tisdall et al., 1986). Management strategies that increase SOM (e.g., reducing tillage increasing soil coverage) also aid in reducing soil erosion, which preferentially removes the SOM-rich top-soil (Cihacek et al., 1993; Gregorich et al., 1998; Lal, 2003).

The vast majority of SOM originates from plant inputs, although this material may pass through several trophic levels prior to acquiring the characteristics of stable SOM. Conversion of plant biomass begins with decomposition; thus, decomposition studies provide insight into early steps of humification. Field and laboratory incubation studies suggest that it is common for 50% or more of the initial plant biomass input to decompose within the first year (Broder and Wagner, 1988; Burgess et al., 2002; Buyanovsky and Wagner, 1997; Johnson et al., 2004; Schomberg et al., 1994; Stott and Martin, 1990). The rate of decomposition in the field depends on climatic conditions (moisture and temperature), particle size, biomass to soil contact, biomass orientation, and plant biochemical composition (Aerts, 1997; Ghidey and Alberts, 1993; Johnson et al., 2007a).

2.3. Charcoal/black C

Charcoal or black C, a unique recalcitrant form of C, is found in many soils, especially those with history of burning activities. In the literature, terminology referring to this type of C includes "charcoal," "char," "bio char," "black C" and "agro-char." In this review, we use "charcoal" as a generic term for this form of recalcitrant C and use "bio char" to specifically refer to biologically active charcoal resulting from biomass pyrolysis. Charcoal results from incomplete combustion (insufficient oxygen) of biomass C (Goldberg, 1985) and can contribute to C sequestration (Fowles, 2007). The physical and chemical properties of charcoal vary tremendously from fly-ash burning to bio char from pyrolysis (Goldberg, 1985). Charcoal is rather ubiquitous in soils, resulting from natural or intentional burning of biomass (Schmidt and Noack, 2000). Charcoal can represent 10—35% of the total SOC and is highly recalcitrant to microbial and chemical decomposition (Skjemstad et al., 2002).

One of the advantages of using bio char as a soil amendment is that C can be locked in the soil for centuries, perpetuating enhanced plant growth and the ability to store and recycle C more efficiently (Fowles, 2007; Lehmann et al., 2006). It has been suggested that converting from "slash and burn" to "slash and char," which is more C and nutrient conservative, could improve soil quality of Oxisols (Lehmann et al., 2002). Adding charcoal in addition to NPK fertilizer improved plant growth and doubled grain yield compared to using inorganic fertilizer alone on a Brazilian Oxisol (Christoph et al., 2007). Bio char has the capacity to reduce CO_2 emissions, making the system C-neutral or in some cases C-negative (Fowles, 2007). Bio char formed under the proper conditions has remarkable nutrient affinity and enhances the cation exchange capacity of soil, as well as biological processes that lead to improved soil structure, water storage, and soil fertility (Fowles, 2007). Bio char can be infused with other nutrients (i.e., N as ammonium bicarbonate) to act as a slow release fertilizer (Day et al., 2002, 2005) and potentially decrease leaching and runoff (Fowles, 2007). Bio char could also adsorb pesticides and other potential pollutants (Lehmann et al., 2006), as well as reduce N_2O and CH_4 emission from agricultural fields (Fowles, 2007). The feedstock and pyrolysis conditions of thermochemical bioenergy platforms can be manipulated to produce bio char. Generation of bio char may require sacrificing some of the energy produced to retain more C sequestration value (Johnson et al., 2007b).

2.4. Soil inorganic carbon (SIC)

Some soils contain inorganic forms of C (i.e., carbonates), which collectively are termed SIC. Especially in arid and semiarid environments, SIC can represent a significant amount of C (Lal, 2002, 2007; Monger and Martinez-Rios, 2001). The SIC pool was estimated as 23–94 Pg C in the 0–2 m profile of the conterminous United States with Mollisols and Aridisols containing the largest portion of the SIC pools (Guo et al., 2006). The SOC pool for this area was estimated as 30–150 Pg C in the 0–2 m profile (Guo et al., 2006).

The impact of management on SIC is just beginning to be explored. Reeder et al. (2004) reported that cattle (*Bos* spp.) grazing had a larger impact on SIC content than on SOC content in the short-grass steppe of Colorado. Heavy grazing increased the abundance of C4 grasses relative to C3 grasses compared to no grazing or light grazing during 56 years. The change in plant community resulted in a shift in the distribution and quantity of root biomass (Reeder et al., 2004). On a mixed-grass prairie in North Dakota, heavy grazing did not increase SIC content, but there was a tendency for increased SIC under fertilized crested wheat grass (*Agropyron desertorum* (Fisch. ex. Link) Schult.) (Liebig et al., 2006). Lal (2007) reviewed the mechanisms involved during the formation of SIC.

2.5. Tillage and residue management

There have been several recent reviews assessing the impact of conservation tillage on soil C sequestration (Franzluebbers and Follett, 2005; Lal, 2007; Six et al., 2004; West and Post, 2002). West and Post (2002) found that a change from conventional tillage to no-tillage (NT) sequestered 0.57 ± 0.14 Mg C ha^{-1} yr^{-1}. Sperow et al. (2003) suggested that adoption of NT on all currently annually cropped area (129 Mha) in the United States could sequester 47 Tg C yr^{-1}. If NT were used on 50% of cropland, and reduced tillage on the remaining 50%, then about 37 Tg C yr^{-1} could be sequestered. The assumptions underlying soil C sequestration with conservation tillage were questioned by Baker et al. (2007), because most studies measured soil C sequestration within the surface of 30 cm only. If sampled below 30 cm, the evidence for soil C sequestration under conservation tillage could be challenged, because of greater variability in detecting differences. Tillage redistributes and mixes plant material and SOC throughout the profile (Staricka et al., 1991). Tillage or the lack there-of had a larger impact on the placement of C rather than the amount of total C in the profile when sampled below 30 cm (West and Marland, 2002). At least in the short-term, tillage induces CO$_2$ emission proportional to the volume of soil disturbed (Reicosky and Archer, 2007; Reicosky and Lindstrom, 1993). Huggins et al. (2007) reported that chisel plow and NT treatments with continuous corn (*Zea mays* L.) contained 164 Mg SOC ha^{-1} (0–45 cm) while moldboard plow corn, and NT continuous soybean (*Glycine max* L. [Merr.]) contained 135 Mg SOC ha^{-1} (0–45 cm). During the course of 14 years, all tillage and cropping treatments lost SOC compared to the initial SOC levels; however, conservation tillage and NT lost the least (Huggins et al., 2007). In contrast, two micrometeorological studies measuring CO$_2$ from conservation-tilled cropland did not find the soil acting as a net sink for C (Baker and Griffis, 2005; Verma et al., 2005). Experiments attempting to document changes in soil C should be sampled below 30 cm, and preferably throughout the rooting zone. In a review of long-term cropping studies, the impact of NT management on C sequestration and GWP was significant at the end of 20 years, but not earlier (Six et al., 2004). Collectively, these results suggest that C sequestration due to NT or conservation/reduced tillage depends on depth of soil sampling, crop management, and duration of continuous low-intensity tillage system.

There are many benefits in reducing tillage beyond C sequestration. No-tillage management is an effective method to control erosion (Gebhardt et al., 1985). In addition to erosion control, NT (Phillips et al., 1980) and reduced tillage (e.g., ridge tillage) also save energy by reducing fossil fuel needs (Archer et al., 2002). West and Marland (2002) noted that enhanced C sequestration due to reduced tillage was finite, but reduced fossil fuel benefits continued as long as reduced tillage was practiced.

2.5.1. Tillage and energy use

In view of rising energy costs (USDA ERS, 2006), the typical operations in traditional agricultural production (including tillage, planting, fertilizing, cultivating, harvesting, processing, transporting, and storage) should be re-examined. Tillage and harvest represent the greatest proportion of fuel consumption within intensive conventional agricultural systems (Frye, 1984). No-till and reduced tillage (e.g., ridge tillage) save energy by reducing fuel needs (Phillips et al., 1980; Archer et al., 2002). Converting from moldboard plowing to NT could save about 20 kg C ha^{-1} yr^{-1} (Frye, 1984). Thus, reduced tillage reduces direct energy consumption and reduces CO$_2$ from fossil fuel combustion.

2.6. Carbon sequestration with land application of animal manures

Consumption of plants by animals ultimately leads to a partitioning of C into animal biomass, CO$_2$ respired by animals, and fecal deposition of C in unutilized co-products. Animal manures in modern, confined agricultural systems are collected to avoid stockpiles and often applied to cropland or grazing land to replenish soil fertility. The fate of animal manures in soil depends upon the environmental conditions, primarily temperature and precipitation, as well as tillage, handling procedures and the method and timing of application (Reicosky et al., 2000). Animal manures are eventually decomposed by soil microorganisms and contribute to the pool of SOM, if stabilized biochemically or biophysically.

Since animal manure contains 40–60% C on a dry weight basis, its application to land should promote SOC sequestration (CAST, 1992). In a review of studies conducted in the southeastern USA, poultry litter application to crop and

pasture lands led to significant change in soil organic C only when evaluations were conducted for more than 2 years (Franzluebbers, 2005). Conversion of C in poultry litter to SOC was $17 \pm 15\%$ among these studies. Although SOC has been shown to increase with animal manure application, very few whole-system data have been collected (Franzluebbers, 2005). Manure application may simply transfer C from one land to another, while investing energy in transport and handling operations. A full C accounting approach is needed for an adequate assessment of manure application as a viable C sequestration strategy.

From a whole-farm simulation of the relative contributions of various agricultural components to SOC in Ontario, Beauchamp and Voroney (1994) concluded "…manure contributes a relatively small proportion of the C to soil compared to crop roots and residue, especially in swine systems, unless straw bedding is included." The small contribution of poultry manure to SOC is supported by a study in Georgia, in which poultry litter applied at rates between 45 and 269 Mg manure $ha^{-1}\,yr^{-1}$ for 2 years resulted in a decline in SOM from loss on ignition and no change in total N (Jackson et al., 1977). However, this interpretation is not supported by several other studies. Long-term data from a corn–bean rotation in Kenya with application of cattle manure increased SOC by an average of 2.1 Mg C $ha^{-1}\,yr^{-1}$, while crop stover retention increased SOC by an average of 1.1 Mg C $ha^{-1}\,yr^{-1}$ during 18 years without inorganic fertilization (Kapkiyai et al., 1999). As a percentage of C applied in manure, 5–12% was retained as SOC, suggesting that significant soil C sequestration (i.e., 0.10–0.23 Mg C $ha^{-1}\,yr^{-1}$) is possible in the humid tropics with repeated and long-term application. The rate of soil organic C sequestration with cattle manure (0.20–0.22 Mg C $ha^{-1}\,yr^{-1}$; $\sim 10\%$ retention of applied C) was similar in a cotton (*Gossypium* spp.)–guinea corn (*Sorghum* spp.)–groundnut (*Arachis hypogaea* L.) rotation conducted for 45 years under the warm, moist climate in Nigeria (Agbenin and Goladi, 1997). In Georgia, SOC sequestration with 5 years of poultry litter applied to bermudagrass (*Cynodon dactylon*) pasture was 0.26 ± 0.55 Mg C $ha^{-1}\,yr^{-1}$ ($14 \pm 30\%$ retention of applied C) under various harvest strategies (Franzluebbers et al., 2001). In Alabama, tall fescue (*Lolium arundinaceum* (Schreb.) Darbyshire) pastures supplied with poultry litter for 21 ± 4 years sequestered SOC at a rate of approximately 0.30 Mg C $ha^{-1}\,yr^{-1}$ (8% retention of applied C) (Kingery et al., 1994).

In the warm, semiarid region of India, application of farmyard manure to a pearl millet (*Pennisetum glaucum* [L.] R. Br.)–wheat (*Triticum aestivum* L.) cropping system for 20 years resulted in a sequestration rate of 5, 4, and 3% of applied manure C with applications of 15, 30, and 45 Mg C $ha^{-1}\,yr^{-1}$, respectively (Gupta et al., 1992). During 56 years of continuous cattle manure application in eastern Oregon, SOC to a depth of 20 cm was 36% greater than with inorganic fertilization, equivalent to a SOC sequestration rate of 0.19 Mg C $ha^{-1}\,yr^{-1}$ ($\sim 23\%$ retention of applied C) (Collins et al., 1992). With 135 years of continuous farmyard manure application in England, SOC sequestration was equivalent to

0.50 Mg C $ha^{-1}\,yr^{-1}$ (17% retention of applied C) (Webster and Goulding, 1989). In the cold, semiarid region of Alberta, cattle manure applied to cropland at selected rates for 11 years nearly doubled SOC of the 0- to 15-cm depth under dryland conditions and more than doubled SOC under irrigated conditions (Sommerfeldt et al., 1988). The rate of SOC sequestration in this study was predominantly a function of the quantity of manure applied, where C retention rates were $32 \pm 4\%$ of applied C, irrespective of water management (Chang et al., 1991). In the cold, moist climate of Quebec, cattle manure applied to corn grown for silage during 10 years resulted in 7–10% retention of applied C in soil, which was equivalent to SOC sequestration rates of 1.5 and 2.2 Mg C $ha^{-1}\,yr^{-1}$ at manure application rates of 40 and 80 Mg $ha^{-1}\,yr^{-1}$, respectively (Angers and N'Dayegamiye, 1991).

In general, SOC sequestration on a land area basis appears to be greater with an increased rate of manure application (Sommerfeldt et al., 1988; Gupta et al., 1992). However, high nutrient loading could pose environmental threats to water quality. Climatic regime is another important variable that appears to affect potential retention of applied C in soil based on the studies cited above. Thermic regions tended to have lower C retention rates from manure ($7 \pm 5\%$) than temperate or frigid regions ($23 \pm 15\%$). Higher temperature would be expected to decompose manure more completely based on thermodynamic mechanisms that control soil microbial activity. Higher moisture would also be expected to accelerate decomposition of applied C to soil, yet C retention rates in moist regions ($8 \pm 4\%$) did not vary significantly from dry regions ($11 \pm 14\%$).

2.7. Impact of organic agriculture

Animal manures and cover cropping in organic agricultural systems can contribute to SOC sequestration. These management practices have implications for production or reduction of CO_2, N_2O, and CH_4. Few data specific on the contribution of organic agriculture to SOC sequestration or GHG emission are available. Organic systems are not necessarily low-input, because they often use intensive tillage for weed control, and use approved organic inputs, and would likely have a - diverse crop rotation relative to conventional systems.

2.7.1. Carbon sequestration with organic agriculture

Management practices that lead to an increase in organic inputs (biomass and manure) enhance microbial functions and promote SOC sequestration (Jarecki and Lal, 2003). Organic production frequently utilizes practices recommended for increasing SOC sequestration such as surface mulching, continuous cropping, cover cropping, legumes in rotation, and manure application. Like organic production other low-input or sustainable management systems incorporating some of the above practices as well as reduced tillage may also provide C sequestration potential.

Research comparing organic practices to conventional practices is limited, especially in the United States. In a comparison of 35 paired organic and conventional farms in the

midwestern United States, Lockeretz et al. (1981) found significantly greater SOC under organic production. In California, an increase of up to 19% SOC was observed in organic and low-input systems compared to a 10% change under conventional production after 8 years (Clark et al., 1998). In Pennsylvania, 22 years of organic practices increased SOM by 15−28%, which was a significant increase over a conventional system that only increased 9% (Pimentel et al., 2005).

Abroad, biodynamic farms established for at least 8 years in New Zealand had greater SOC levels than conventional farms (Reganold et al., 1993). In southeastern Australia, particulate organic C accumulated under organic farming, (Chan, 2001). In the Netherlands, SOM levels were significantly greater under organic farming compared to conventional farming after 70 years of management (Pulleman et al., 2000, 2003). Reports from Denmark and Belgium also suggest that organic production systems have the potential to sequester as much as 0.5 Mg C ha^{-1} yr^{-1} depending on the specific management practices utilized (Dendoncker et al., 2004; Foereid and Høgh-Jensen, 2004; Freibauer et al., 2004). Lal (2004a) identified practices to improve land management in central Asia and northern Africa including organic farming, which through manure applications and incorporation of legumes into the rotation could sequester SOC. Freibauer et al. (2004) estimated global C sequestration potential of 0.1−0.8 Mg C ha^{-1} yr^{-1} by conversion to organic management. Similar global estimates (0.3−0.6 Mg C ha^{-1} yr^{-1}) were made by Pretty and Ball (2001). A study by Smith et al. (2005) indicated that organic farming is a promising management system for enhancing C storage on cropland.

2.7.2. Energy considerations with organic agriculture

All management systems require energy expenditures. Analysis of total direct and indirect energy consumption is one method to compare systems. Direct consumption is linked to field operations including tillage, amendment application (e.g., manure, fertilizer, etc.) and harvest. Indirect consumption is linked to the production of amendments, and transportation and storage of these and other products to and from the field (Shepherd et al., 2003; Stolze et al., 2000; Storey, 1997). Direct energy use was 60% lower for an organic system compared to a conventional system, primarily through reduced field traffic (Cobb et al., 1999). Total energy consumption with organic agricultural systems can be substantially lower than intensive conventional agricultural systems, other conservation or sustainable practices can also reduce total energy consumption (Pimentel et al., 2005; Pretty and Ball, 2001; Shepherd et al., 2003; Stolze et al., 2000). On a per unit economic value Lockeretz et al. (1981) calculated that organic systems required 40% of the fossil energy of conventional systems. In contrast, some studies and reviews indicate that energy consumption based on units of output, or production units, showed reduced benefit of the organic system for energy efficiency (Shepherd et al., 2003; Stolze et al., 2000).

Switching to management practices that improve N use efficiency and pest management may reduce or eliminate direct and indirect energy consumption (Storey, 1997). A diverse crop rotation, particularly one that includes legumes, typically enhances fertilizer-use efficiency and improves pest management (Jarecki and Lal, 2003; Pimentel et al., 2005; Tillman et al., 2004). A diverse crop rotation has the potential to lower application rates or frequencies (direct), resulting in less direct (application cost) and indirect (chemical production and transportation) energy consumption related to chemical amendments. An off-farm strategy to reduce fuel consumption would be to move to a more regionally based economy, in which there would be reduced distance between producer and consumer.

3. Methane

Agriculturally produced methane is emitted biologically to the atmosphere via methanogenic bacteria living in anaerobic soils (i.e., flooded rice patties), the rumen of cattle and sheep (*Ovis* spp.) and in the intestinal tract of termites. Methane contributes about 20% of the estimated anthropogenic radiative forcing, second behind CO_2 at 60% (Lassey, 2007). In 2000 concentration of CH_4 in the atmosphere was 1750 ppb, which far exceeded the concentration ($\sim$700 ppb) in 1700 AD (Etheridge et al., 1998). US EPA (2007) estimated that about 28% of CH_4 emissions was the result of livestock products; second only to landfill emission (Fig. 2). Livestock production of CH_4 (i.e., both from enteric fermentation and from animal waste storage) was estimated to be 20−34% of all CH_4 produced globally (CAST, 1992; IPCC, 2000). Global sources of CH_4 are livestock production, rice farming, waste decomposition (animal waste, crop residues, and landfills), and fossil fuel mining. Current global atmospheric content of CH_4 was estimated as 4900 Tg (Lassey, 2007).

3.1. Impact of animals and animal by-product management

Generally, livestock farming systems can be categorized into three broad systems: grazed, mixed, and landless. They convert plant carbohydrates and protein into animal protein with an efficiency of 5−45%, depending on animal type, system, and management (Oenema, 2006). Ruminant livestock (cattle, buffalo (*Bison* spp.), sheep, goat (*Capra* spp.), camel (*Camelus* spp.), deer (*Odocoileus* spp.), etc.) are a major source of CH_4. Manure from all livestock (ruminants, as well as other major livestock, including poultry and swine (*Sus* spp.)) is a contributor to N_2O emission (US EPA, 2005). Combined with CO_2 emission via animal respiration and manure decomposition, these trace gases are of prime significance to understanding the impact of animals on global greenhouse gas emissions.

Livestock sources of CH_4 are predominantly enteric (i.e., from the breath of ruminants and flatus of monogastric animals) as a result of feed digestion. Globally, enteric production of CH_4 was estimated as $\sim$80 Tg yr^{-1}, which was 20−25% of the observed increase in atmospheric CH_4 concentration (Lassey, 2007). Additionally, CH_4 is produced from anaerobic decomposition of animal manure in slurry pits, solid manure piles, and

from wet soil following incorporation of manure (Lassey, 2007).

The relative contribution of enteric versus manure sources of CH_4 is dependent upon animal type (Table 2). Monteny et al. (2006) outlined some of the factors affecting CH_4 production from ruminant livestock production, which included level of feed intake, quantity of energy consumed and feed composition. Therefore, livestock management impacts the quantity of CH_4 produced. Three important factors play a role in modifying enteric CH_4 production (rate of organic matter fermentation, type of volatile fatty acids produced, and efficiency of microbial biosynthesis) (Monteny et al., 2006). Total CH_4 production increases with greater feed intake, but the proportion of gross energy consumed and converted to CH_4 is reduced. By reducing rumen pH, H^+ available for CH_4 formation becomes limiting. Common ionophores, such as monensin, can alter digestive microflora to reduce CH_4 production by as much as 30% (Guan et al., 2006). Compared to a 9% reduction in CH_4 with ionophore supplementation, a 21% reduction in CH_4 production was found with fatty acid supplementation (sunflower (*Helianthus annus* L.) oil) (McGinn et al., 2004). Under a high-grain diet, cattle produced less CH_4 when fed corn (2.8% of gross energy consumed) than when fed barley (*Hordeum vulgare* L.) (4.0% of gross energy consumed), both of which were less than when fed a low-grain diet (7.4% CH_4 from gross energy consumed) (Beauchemin and McGinn, 2005). From undisturbed measurements in Australia, cattle grazing poor quality pasture produced ~8% CH_4 from gross energy consumed (0.23 kg CH_4 animal^{-1} d^{-1}), while cattle fed a high-grain diet produced ~2% CH_4 from gross energy consumed (Harper et al., 1999).

Monteny et al. (2006) calculated that enteric fermentation from swine might generate up to 2% CH_4 of the gross energy consumed compared with 7% in ruminants. Methane emission from swine lagoons in North Carolina was 8–62 kg CH_4 ha^{-1}, equivalent to 1.6–6.0 kg CH_4 animal^{-1} yr^{-1} (Sharpe et al., 2002). In a series of four lagoons designed to successively purify water from a swine production facility, the gas flux from Lagoon 1, which directly received animal waste was dominated by CH_4 (79% of total gas flux) with smaller quantities of dinitrogen (N_2) (15%) and N_2O (<1%) (Harper et al., 2000). The gas flux from Lagoons 2, 3, and 4 was dominated by N_2 (54, 59, and 69%, respectively) with smaller quantities of CH_4 (26, 13, and 8%, respectively) and N_2O (<1, 3, and 18%, respectively). Methane and N_2O flux were inversely related to dissolved oxygen in the lagoons (1, 8, 19, and 20 mg L^{-1} in Lagoons 1, 2, 3, and 4, respectively). A portion of CH_4 emitted to the atmosphere can be sequestered by

aerobic soils. Thus, land application of manure could significantly decrease the net quantity of CH_4 emitted to the atmosphere compared with stockpiling or long-term lagoon storage of manure. Manure applied to pasture land did not appear to impact CH_4 emission (Chadwick et al., 2000). In contrast, net uptake of CH_4 under corn from moldboard plowed soils amended with manure was reduced relative to soils without manure (Rochette and Cote, 2000).

3.1.1. Impact of organic animal production systems

In an organic system there is greater potential for the production of CH_4 from livestock because animals tend to ingest a relatively low quality forage diet compared to cows in grain-fed systems (Phetteplace et al., 2001; Stolze et al., 2000). Comparisons of organic and conventional livestock production indicate that CH_4 emissions are lower for organic systems when based on a unit of area (Cobb et al., 1999); however, per production unit particularly with dairy there were little to no differences between the two systems (Shepherd et al., 2003; Sneath et al., 2006; Stolze et al., 2000). There is a large effect on CH_4 emissions that is contingent on the production and use of farmyard manure. Typically, in an organic system, stock piled manure is composted, which will increase aeration limiting anaerobic production of CH_4. In addition, livestock production under organic practices incorporates a pasture component as well as straw-based housing. Conventional production systems often use lagoon systems for manure handling, which produce large amounts of CH_4 under anaerobic conditions (Harper et al., 2000).

3.2. Soil and methane

Soils can either be a net sink or source of CH_4, depending on moisture, N level and ecosystem (Chan and Parkin, 2001a; Gregorich et al., 2005; Liebig et al., 2005). Methane from rice production contributes about 1% of the CH_4 in the Untied States (Fig. 2), while globally it accounts for about 20% (IPCC, 1992). Methane is consumed by soil methanotrophic bacteria, which are ubiquitous in many soils (McLain and Martens, 2006) but is also produced by methanogenic bacteria (Chan and Parkin, 2001a). Native prairie and forest systems tend to be net CH_4 consumers, while agricultural systems (without manure) can be consumers, producers or neutral depending on the growing season (Chan and Parkin, 2001a).

3.2.1. Rice and methane

Globally, most rice is grown and consumed in Asia (Wassmann et al., 2000a). Methane from rice production varies substantially among watering strategies. Globally, irrigated rice accounts for 70–80%, rainfed for 15% and deepwater rice for about 10% of the CH_4 produced from rice (Wassmann et al., 2000b). Upland rice is not considered a significant source of CH_4 (Neue, 1997). Water management during the production of rice is a key factor in minimizing CH_4 during rice production. Draining the water and allowing the soil to become aerobic allows oxidation of CH_4 and reduces CH_4 production (US EPA, 2006). A review by Reicosky et al. (2000) summarized other

Table 2
Typical distribution of CH_4 emission from enteric fermentation among animal types (Monteny et al., 2001)

Animal type	Total CH_4 (kg CH_4 animal^{-1} year^{-1})	Total CH_4 as enteric fermentation (%)
Dairy cattle	84–123	75–83
Swine	4.8	30
Poultry	0.26	~0

mitigation strategies including fertilizer and manure management, choice of cultivar and biomass (straw) management. Sulfur containing fertilizers can increase the competitive advantage of sulfate reducing bacteria relative to methanogens, thus reducing CH_4 production (Van der Gon et al., 2001).

About 65% of the United States' rice is produced in Southeastern states (i.e., Arkansas, Louisiana, Missouri and Mississippi), although CH_4 emission data are scarce (Franzluebbers, 2005). Several reports are available that document similar mechanisms to those summarized globally. Lindau et al. (1991) measured increased CH_4 emission when urea application was >200 kg ha^{-1} in flooded rice fields of Louisiana. Methane production from rice was greater when rice-straw was incorporated than when removed or burned (Bossio et al., 1999; Redeker et al., 2000). Semi-dwarf cultivars grown in Louisiana emitted significantly less CH_4 compared to tall varieties (Lindau et al., 1995).

3.2.2. Consumption of CH₄ by soil

Methane consumption averaged 1.4 ± 0.84 kg CH_4-C ha^{-1} yr^{-1} ($n = 18$) across the dryland systems in the Great Plains (Liebig et al., 2005), 0.3 ± 0.3 kg CH_4-C ha^{-1} yr^{-1} ($n = 20$) in eastern Canada (Gregorich et al., 2005) and about 1.2 kg CH_4-C ha^{-1} yr^{-1} in southwestern United States (Martens et al., 2005). In Iowa, both production and consumption were reported from agricultural lands, but prairie soils and forest soils consumed 1.9–4.1 kg CH_4-C ha^{-1} yr^{-1} (Chan and Parkin, 2001a). Application of inorganic fertilizer tends to decrease CH_4 consumption (Bronson and Mosier, 1993; Chan and Parkin, 2001b).

Soils have the potential to consume CH_4; the question is, how much can be consumed relative to animal production. Assuming consumption rates of 0.4–1.9 kg CH_4 ha^{-1} yr^{-1} (Gregorich et al., 2005; Liebig et al., 2005); it would take 45–213 ha of soil to offset an enteric CH_4 production of one animal producing 83 kg CH_4 animal^{-1} yr^{-1} (Table 2). The 97 million head of cattle in the United States (February 2007 inventory, http://usda.mannlib.cornell.edu/usda/current/USCC/USCC-02-15-2007.pdf) would produce about 8.1 Tg enteric CH_4 annually. Typical uptake of CH_4 by soil could only offset 4.5–21% of the annual enteric CH_4 emission by cattle. Gregorich et al. (2005) reached a similar conclusion estimating that uptake by agricultural lands in eastern Canada could only offset about 4% of the CH_4 produced by dairy cows in the region.

4. Nitrous oxide emission and remediation

Sixty-five percent of all N_2O emission is estimated from soil microbial processes of nitrification (aerobic transformation of ammonium to nitrate) and denitrification (anaerobic transformation of nitrate to N_2 gas) (Monteny et al., 2006; Smith and Conen, 2004). The main cause of agricultural increases in N_2O emission to the atmosphere is the application of N fertilizers and animal manures (Fig. 3) (Storey, 1997).

4.1. Impact of animals and animal manure management

Global production of N in animal manure has been estimated as 94 Tg (Oenema, 2006), which is of similar magnitude to that of fertilizer N produced each year (70–80 Tg, Fixen and West, 2002). Cattle account for 56% of the manure N, while swine accounts for 11% and poultry accounts for 9% (Oenema, 2006). About half of the manure N is collected and stored and half is deposited directly on the land. Management opportunities are available to manipulate the amount of manure N emitted as N_2O.

Potential denitrification is strongly influenced by the water-soluble organic C content of manures, with slurries often containing greater concentrations of this important component than stacked or composted manures (Paul and Beauchamp, 1989). A soil with three times greater soil organic C due to long-term manure application resulted in 29 kg N ha^{-1} lost via denitrification during the October–November period following barley harvest compared with 5 kg N ha^{-1} lost in soil without manure application (Webster and Goulding, 1989). Liquid manure provides readily oxidizable C and sufficient mineralizable N to activate the population of denitrifiers in soil. Comfort et al. (1990) found that most of the N_2O emission occurred within the first 5 days following injection of manure into the soil when soil CO_2 evolution was greatest. Rainfall that saturated soil at 25 days following injection resulted in little emission of N_2O despite the presence of significant nitrate, due to much lower microbial activity following exhaustion of readily decomposable C. Rapid, but relatively brief (i.e., within 30 days) emission of N_2O occurred following swine slurry application to a soil with a history of manure application in Quebec (Rochette et al., 2000). Cumulative N_2O-N loss was 1 kg ha^{-1} yr^{-1} (0.6% of total N applied) from inorganically fertilized soil, 2 kg ha^{-1} yr^{-1} (1.2% of total N applied) from soil receiving 60 Mg ha^{-1} yr^{-1} of swine slurry, and 4 kg ha^{-1} yr^{-1} (1.7% of total N applied) from soil receiving 120 Mg ha^{-1} yr^{-1} of swine slurry.

Rotz (2004) outlined several management options to reduce N loss from livestock production. Management should focus on improving N-use efficiency of animals to reduce N excretion, retaining that N contained in manure until it is applied to land, and applying the appropriate amount of manure in a timely manner to enhance crop uptake. Manure slurry ponds are typically a significant source of CH_4, but N_2O emission can be important when ponds become encrusted or are covered (Rotz, 2004). Berg et al. (2006) found that lowering the pH of slurry ponds is effective in reducing both CH_4 and N_2O emissions. Cattle over-wintering areas can be a source of significant N_2O emission as a result of a large accumulation of excrement, high nitrate concentration, and wet soil conditions (Hynšt et al., 2007). Whether volatile loss of N from stored manure is in the form of ammonia or N_2O depends on how manure is stored and subsequently applied to land (Thorman et al., 2006).

Nitrous oxide emission from livestock feces deposited on pasture is dependent on rainfall, quantity and frequency of N-inputs from stocking rate, and soil organic C level (Saggar et al., 2007). From a balance of CO_2, CH_4, and N_2O emissions

at nine sites in Europe, grasslands were a sink for atmospheric CO_2 (2.40 Mg C ha^{-1} yr^{-1}), which was only marginally offset by emission of CH_4 (0.32 Mg CO_2-C equivalent ha^{-1} yr^{-1}) from enteric fermentation of grazing cattle and emission of N_2O (0.14 Mg CO_2-C equivalent ha^{-1} yr^{-1}) from nitrification and denitrification in soil (Soussana et al., 2007). Accounting for the off-site emissions of CO_2 and CH_4 from harvested and fed forage, the sink capacity for GHG balance was reduced further. In Ireland, two dairy farms were found to be sinks for GHG balance at the rate of $\sim$0.3 Mg CO_2-C equivalent ha^{-1} yr^{-1}, primarily due to the high fixation of CO_2 in forage (Byrne et al., 2007).

4.2. Effects of management on N_2O emission

In general, N_2O emission increases with increased N-inputs (Gregorich et al., 2005; IPCC, 2001). The proportion of applied N emitted as N_2O (independent of source of N) has been estimated as 1.25% (IPCC, 1997). A positive relationship was reported between N fertilizer applied and N_2O flux ($r^2 = 0.42$) in Eastern Canada (Gregorich et al., 2005). In contrast, annual N_2O emission and N fertilization were poorly correlated on cropland when comparing among reports from many crops and management systems (Roelandt et al., 2005). The apparent lack of correlation may be due in part to the nonlinear relationship between N-inputs and N_2O flux. McSwiney and Robertson (2005) reported that N_2O fluxes were low to moderate until the N-input exceeded crop needs, after which the flux nearly doubled; suggesting that prudent management of N-inputs can be an effective strategy to minimize N_2O emitted from agricultural lands. Emission of N_2O after application of anhydrous ammonia was 2–4 times greater than surface applying urea ammonium nitrate or broadcasting urea (Thornton et al., 1996; Venterea et al., 2005). Fertilizing with urea increased the potential for ammonia emission, which can contribute to acidification and eutrophication during subsequent deposition compared to fertilizing with ammonium nitrate (Harrison and Webb, 2001).

As N_2O emissions are strongly related to N-inputs, it has been suggested that emissions may be greater from manure sources rather than conventional fertilizers (Peterson et al., 2006). From a review of studies in eastern Canada, the proportion of applied N emitted as N_2O averaged $1.7 \pm 1.0\%$ ($n = 12$) with liquid animal manure, $0.3 \pm 0.1\%$ ($n = 4$) with solid animal manure, $5 \pm 13\%$ ($n = 24$) with inorganic N to annual cropland, and $1.2 \pm 2.2\%$ ($n = 9$) with inorganic N to perennial crops (Gregorich et al., 2005). Liebig et al. (2006) found a positive and highly significant relationship ($r^2 = 0.85$) between soil particulate organic matter and N_2O flux from grazed pasture in the northern Great Plains. The proportion of applied N emitted as N_2O (independent of source of N) has been assigned a base value of 1.25% (IPCC, 1997).

Substantial N_2O emission can occur during freeze–thaw events (Chang and Hao, 2001; Gregorich et al., 2005; Muller et al., 2002). Even though soil temperatures may be near 0 °C, the emission of N_2O is due to microbial activity (Chang and Hao, 2001) and not simply a physical release of entrapped

gas (Sharma et al., 2006). Release of organic C from plants also appears to play a role in providing C substrate necessary for the production of N_2O, which is primarily due to denitrification rather than nitrification (Morkved et al., 2006). Furthermore, the production of N_2O exceeds its reduction to N_2 at low temperature, thus contributing to N_2O emission during spring-thaw events (Holtan-Hartwig et al., 2002).

The information available on N_2O emission varies considerably among regions in the United States. In the northwestern United States and western Canada, N_2O emission ranged from 1.0 to 7.1 g N_2O-N ha^{-1} d^{-1} ($n = 28$) under dryland cropping and 1.2–24.5 g N_2O-N ha^{-1} d^{-1} ($n = 13$) with irrigation (Liebig et al., 2005). Rates were highly variable and interacted with crop and fertilizer form and rate. In eastern Canada, emissions averaged 1.2 ± 1.3 kg N_2O-N ha^{-1} ($n = 23$) during spring-thaw events, 2.7 ± 2.2 kg N_2O-N ha^{-1} yr^{-1} ($n = 67$) for all crops receiving N fertilizer, and 1.5 ± 1.0 kg N_2O-N ha^{-1} yr^{-1} ($n = 14$) from corn without N fertilizer (Gregorich et al., 2005). Conservation tillage treatments averaged 7.3 ± 14.3 N_2O-N ha^{-1} yr^{-1} and moldboard plow treatments averaged 3.6 ± 5.6 N_2O-N ha^{-1} yr^{-1} ($n = 15$). Emissions were skewed by very high rates observed on clay soils and this suggests N_2O mission was regulated by soil aeration (Gregorich et al., 2005).

There are a few reports of N_2O emission from other areas of North America. In Iowa during 2 years, N_2O emission from soil under soybean was 4.4–5.3 kg N_2O-N ha^{-1} yr^{-1} and from soil under corn was 9.6–11.5 kg N_2O-N ha^{-1} yr^{-1} (Parkin and Kaspar, 2006). In Wisconsin during two summers, N_2O emission was 2.5–3.3 kg N_2O-N ha^{-1} under corn and 0.9–1.8 kg N_2O-N ha^{-1} under tobacco (*Nicotiana tabacum* L.) (Goodroad et al., 1984). During summer in New York, N_2O emission under corn was 0.3 kg N_2O-N ha^{-1} without N fertilizer or with calcium nitrate and 2.5 kg N_2O-N ha^{-1} when fertilized with urea (Duxbury and McConnaughey, 1986). During summer in Tennessee, N_2O emission under corn was 1.4 kg N_2O-N ha^{-1} without N fertilizer, 13.8 kg N_2O-N ha^{-1} with anhydrous ammonia, and 7.8 kg N_2O-N ha^{-1} when fertilized with urea (Duxbury and McConnaughey, 1986). In Minnesota and Michigan, N_2O emission ranged from 2.4 to 52.2 g N_2O-N ha^{-1} d^{-1} under corn with rates varying by rate and form of N applied (McSwiney and Robertson, 2005; Venterea et al., 2005). The relative scarcity of data and the inconsistency in how data are reported makes it difficult to assess overall impacts of crop and soil management on N_2O emission rates.

4.3. Effects of organic management on N_2O emission

N_2O production potential in organic systems depends upon tillage, manure applications, legume crops (source of N) and incorporation of pasture/grasslands for organic livestock production. Syväsalo et al. (2006) showed that N_2O fluxes were lower under organic grass production than conventional cereal production, but were not significantly different from conventional grass production. Under organic management, Cobb et al. (1999) and Petersen et al. (2006) calculated lower

emissions of N_2O. Evaluations of N_2O emission based on land area were lower under organic than conventional agriculture, but when based on equivalent production, N_2O emission was similar between production systems (Cassman et al., 2003; Shepherd et al., 2003; Stolze et al., 2000).

5. Trade-offs and GWP

Lal (2007) estimated that SOC sequestration potentially could offset about 15% of the global CO_2 emission. There are potential trade-offs between SOC sequestration with conservation tillage enhanced N_2O emission. Reducing tillage to sequester C may increase the frequency or duration of denitrifying condition in soil because the soil remains wetter and additional N fertilizer may be used. As little as 2.5 kg $N_2O\,ha^{-1}\,yr^{-1}$ could offset an annual C sequestration rate of 0.3 Mg C ha^{-1}, because of differences in GWP between CO_2 and N_2O (Johnson et al., 2005). Greater fluxes of N_2O are expected under NT than under conventional tillage (Freibauer et al., 2004; Dendoncker et al., 2004). In a review of 12 studies around the globe, NT systems generally emitted more N_2O than conventional tillage systems, which could reduce the ability of NT systems to mitigate GWP (Smith and Conen, 2004). Other literature reviews (Gregorich et al., 2005; Liebig et al., 2005) did not find a consistent relationship between tillage and N_2O emission with both increased and decreased N_2O emission comparing NT to conventional tillage. In Iowa and Minnesota, N_2O emission from NT was not necessarily greater than from conventional tillage (Parkin and Kaspar, 2006; Venterea et al., 2005). The interaction of soil, climate and management systems needs further evaluation.

5.1. Net GWP

So far in this report we have focused primarily on GHG emission; however, assessing the impact of agriculture on global climate change requires converting emission data to GWP. Cole et al. (1997) estimated that agriculture has the potential to reduce radiative forcing from 1.2 to 3.3 Pg CO_2-C equivalents yr^{-1}. They estimated that about 32% could be from reduced CO_2 emissions, 42% from C offsets through biofuel production on 15% of the existing croplands, 16% from reduced CH_4 emission and 10% from reduced emission of N_2O.

Net GWP calculations should take into consideration the sum of GHG emission, biomass production and ideally net changes in SOC. Mosier et al. (2005) reported that SOC storage relative to total emission determined whether a site would provide a net increase or decrease in GWP. Their comparisons included GWP for farm operations (planting, harvesting, and applying pesticides), fertilizer, liming, irrigation, N_2O, CH_4 and change in surface SOC (0–5 or 0–7.5 cm). Scenarios with fallow or conventional tillage had a net increase in GWP for sites in Colorado and Michigan sites. In Michigan, NT had a net increase in GWP; but it was an order of magnitude less than with conventional tillage. The only scenario in Michigan to decrease GWP was alfalfa (*Medicago sativa L.*).

This was consistent with a previous report from Michigan where systems with perennial species had a net decrease in GWP (Robertson et al., 2000). At an irrigated site in Colorado, NT systems decreased GWP as did elimination of fallow at a Colorado dryland site (Mosier et al., 2005). Six et al. (2004) estimated greater reduction in GWP over a 20-year time span from wetter climates more so than in dry climates.

Calculating net GWP appears straightforward; sum the GHG emission from all sources (soil, energy use, etc.) and compare with the sum of total GHG consumption (C sequestration, methane consumption, etc.). However, measurements are fraught with high spatial and temporal variability. Mosier et al. (2006) compared two methods of estimating net GWP, one based on SOC change (0–7.5 cm) and the other based on soil respiration. The two calculation methods resulted in very different estimates of GWP, both in magnitude and direction. One possible cause for the divergent results may have been caused by accounting for SOC changes only in the surface 0–7.5 cm, especially in light of the argument presented by Baker et al. (2007) concerning shallow versus deep sampling depth. However, annual respiration rates (even micrometeorological techniques) do not give an indication of long-term soil C sequestration, since they only measure short-term net exchange. It is important that when reporting GWP that assumptions and calculations are carefully and clearly delineated.

5.2. Bioenergy

The energy sector is the largest contributor to GHG emission (Fig. 1); therefore, it is essential to curb consumption of fossil fuels if progress is to be made in curtailing GHG emission. Carbon-neutral (causing no net change in atmospheric C concentration) and/or C-negative (causing atmospheric C concentration to decline) fuels are needed. Numerous energy alternatives to fossil fuels exist (e.g., biomass, solar, geothermal, wind, ocean thermal, and tidal), which currently represent only a small fraction of the global energy used (Hoffert et al., 2002). There is ample solar energy; however, current costs of recovery and conversion are high relative to current fossil fuels (Crabtree and Lewis, 2007). Biomass, similar to other renewable energy sources, has a low power density (about 0.6 W m^{-2}) and that alone would not be expected to contribute significantly to climate stabilization (Hoffert et al., 2002). However, biomass can be used to produce C-neutral fuels to power the transportation industry (Hoffert et al., 2002). Biomass fuels are C-neutral because it releases recently fixed CO_2, which does not shift the C-cycle. However, the potential for biomass pyrolysis to produce liquid biofuels and a bio char co-product returned to the soil can be C-negative (Fowles, 2007).

There has been a lively debate in the literature on the pros and cons of bioenergy – from corn grain, cellulosic ethanol fermentation, biomass gasification, and biodiesel (Farrell et al., 2006; Hill et al., 2006; Pimentel and Patzek, 2005; Shapouri et al., 2003; Tilman et al., 2006). The debate focused on valid concerns about energy efficiency, environmental ramifications and potential competition of food crops (e.g., corn and soybean)

for bioenergy feedstock (Hill et al., 2006). Energy balance estimates for grain ethanol production range from negative (Pimentel and Patzek, 2005) to positive (Kim and Dale, 2005; Shapouri et al., 2003). Farrell et al. (2006) concluded grain ethanol could have a small positive energy balance, such that 5–26% of the energy could be considered truly renewable and reduce GWP by about 13%, with considerably greater reduction in GWP with cellulosic ethanol. Soydiesel in the United States is estimated to yield a net energy return of about 93% (Hill et al., 2006). The higher energy return for soydiesel is partly due to the greater energy content of the fuel, but largely due to lower inputs for feedstock production (e.g., fertilizer inputs, energy, and equipment).

Technology is rapidly advancing to utilize crop biomass, perennial grasses, woody perennials and forest products for the production of ethanol via a cellulosic platform and/or utilizing pyrolysis to generate syngas and other products/co-products (Johnson et al., 2007b). As the new technology develops, it is important to avoid negative environmental consequences such as accelerated soil erosion or loss of SOC. Many estimates on the amount of crop biomass that can be harvested are based primarily on erosion risks (e.g., Graham et al., 2007; Nelson et al., 2004; Perlack et al., 2005) with limited estimates that consider the amount of biomass needed to prevent loss of SOC (Johnson et al., 2006). As has been noted by several researchers (Lal, 2004b; Larson, 1979; Wilhelm et al., 2004), crop non-grain biomass is valuable for the land; thus a conservative approach is necessary to develop sustainable biomass harvest rates. Johnson et al. (2007b) concluded that (1) biomass energy feedstock should come first from material diverted from landfills (e.g., bagasse, culled fruits and vegetables, food processing wastes, saw dust and used vegetable oils); (2) agricultural biomass (e.g., straw and stover) should only be harvested once the needs for protecting the soil from wind and water erosion and loss of SOC have been met; (3) feedstocks (annual and/or perennial) need to be regionally developed to meet local needs; and (4) management strategies are needed to assure that the soil resource does not lose its ability to provide food, feed, fiber, and fuel.

6. Research needs and conclusions

Although GHG emission derived from soil has been researched for several decades, there are still geographic regions and agricultural systems that have not been well characterized. There is a need to estimate GWP across a wide range of agricultural systems. Ideally, a standard or conventional method of calculating GWP should be established. Methodology to improve the accuracy of determining changes in SOC and GHG emissions would reduce the uncertainty of estimating GWP.

Agricultural practices that promote good land stewardship appear to minimize or reduce GWP. Such practices include (1) reducing tillage, which helps prevent soil erosion and has the potential to increase SOC, and may enhance CH_4 consumption, even though in some instances it may be partially offset by N_2O emission; (2) eliminating fallow and keeping the soil covered with residue, cover crops or perennial vegetation, which have the potential to increase SOC; (3) avoiding over application and using split N application rates to meet plant need, which would reduce N_2O emission and minimize potential water degradation; and (4) manipulating animal diet and manure management practices to reduce CH_4 and N_2O emission. Opportunities exist in all segments of agriculture to reduce environmental impacts.

As a society, we (farmer/producers and consumers) need to take ownership of the anthropogenic impacts we are having on the environment to avoid disastrous climate change. Conservation programs need to be integrated and linked to C management. Policies that support best agricultural land management practices and land stewardship rather than promoting production of specific crops are needed. Furthermore, concerted efforts are needed to curb fossil fuel demand in the agricultural sector and throughout society. Agriculture has the potential to reduce its environmental footprint and offset GHG emissions. Burning fossil fuel is by far the greatest contributor to GHG emission. We need to develop sustainable alternatives to fossil fuels to reduce our energy demand, thereby, stabilizing GHG emissions and minimizing anticipated global climate change.

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