Chapter 16
Strategies for Mitigating Greenhouse Gas Emissions from Agricultural Ecosystems

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Abstract Climate change, driven by rising greenhouse gas (GHG) concentrations in the atmosphere, poses serious and wide-ranging threats to human societies and natural ecosystems all over the world. Agriculture and forestry account for roughly one-third of global emissions, including 9 to 14% of GHGs from crop and livestock activities. Due to increasing demand based on human population and income growth and dietary change, GHG emissions are likely to increase by about 76% by 2050 relative to the levels in 1995. Nitrous oxide (N_2O) and methane (CH_4) are the major GHGs contributed from the agricultural sector, contributing 50 and 70%, respectively, to the total levels. However, carbon dioxide (CO_2) emissions are mainly contributed by a change in land use patterns and decomposition of organic materials. Global emission pathways that would limit warming to 1.5 °C or less, in line with the Paris Agreement's temperature goal, depend on significant reductions in agricultural GHGs (N_2O and CH_4) as well as net zero CO_2 emissions from fossil fuels. As the agricultural sector mainly contributes to N_2O and CH_4, 4.8 Gt CO_2-eq reduction in direct global agricultural non-CO_2 emissions below baseline by 2050 is needed. These ambitious targets of mitigation pathways present an enormous challenge, and accomplishment of these goals is only possible by the implementation of effective GHG mitigation strategies to the agricultural sector. Mitigation measures in the agricultural sector include increasing C sequestration as well as reduction in the GHGs from livestock and agricultural processes. In this chapter, we discussed mitigation strategies for GHG emissions from the agricultural sector at the global scale.

Keywords Agricultural systems · Carbon sequestration · Climate change · Ecosystems · Greenhouse gas · Livestock

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16.1 Introduction

Over the last several decades, an increase in agricultural GHG emissions has been reported, along with the growing global agricultural production. Agriculture and forestry, which together account for roughly one-third of global emissions, have received much attention in recent years. According to the Intergovernmental Panel on Climate Change (IPCC 2014), the agricultural sector is the second highest GHG contributor after electricity and heat production sector, with this last sector contributing about 24% of the global GHG emissions. However, crop and livestock production are expected to increase by 48% and 80% by 2050, respectively, as the human population grows and shifts toward a more animal-based diet (Bennetzen et al. 2016). Thereby, this scenario for increasing crop and livestock production poses the risk to increase by 76% in agricultural GHG emissions by 2050 relative to 1995 (Popp et al. 2010). The global trends in total GHG emissions from agriculture, forestry, and other land use activities between 1970 and 2010 are presented in Fig. 16.1 (Smith et al. 2014). According to GlobAgri-WRR model, it is projected that GHG from the agricultural sector alone would fill about 70% of the allowable "emissions budget" in 2050 (15 of 21 Gt), leaving almost no space for emissions from other economic sectors and making the achievement of even the 2 °C target

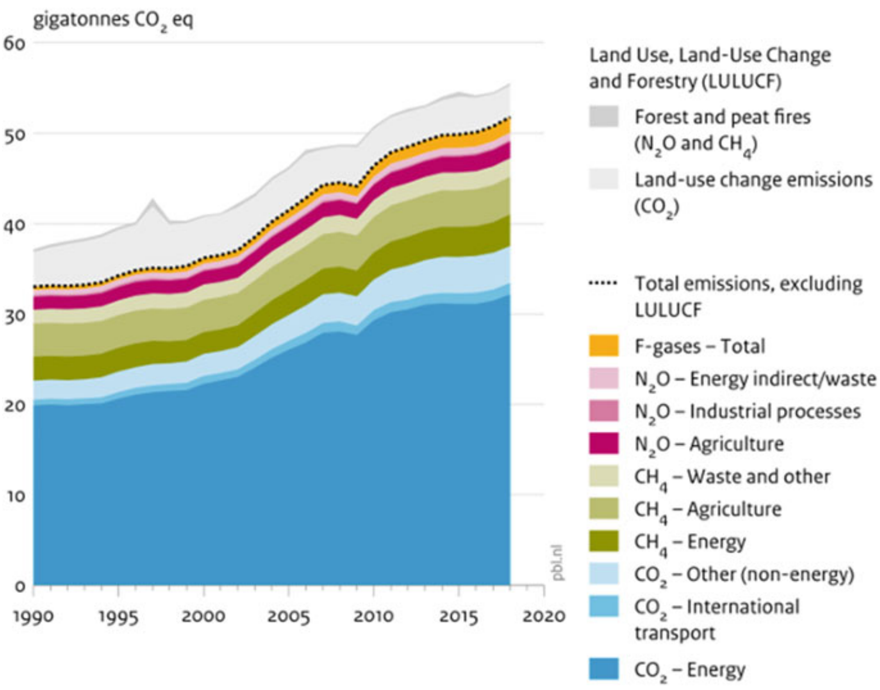

Fig. 16.1 Global trends in total GHG emissions from agriculture, forestry, and other land use activities between 1990 and 2018. (Adapted from Olivier and Peters 2020)

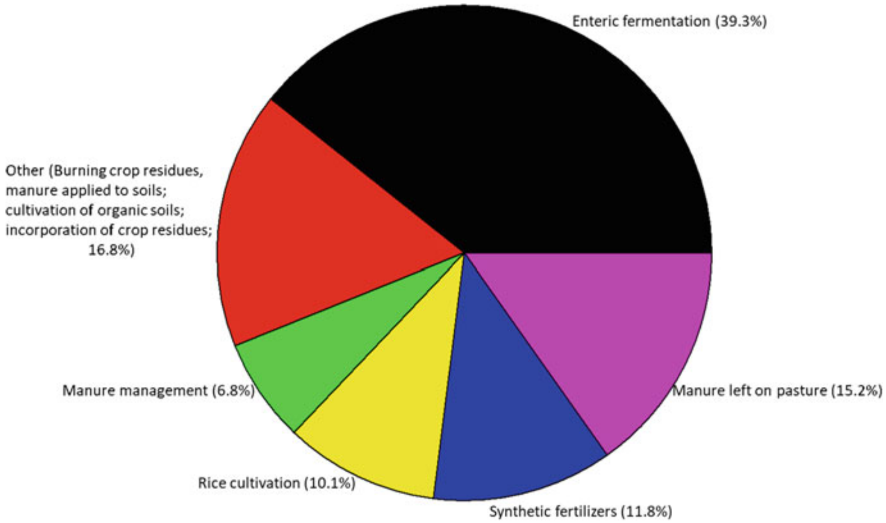

Fig. 16.2 Different sources responsible for these agricultural GHG emissions and their percent contribution. (Source: Data adapted from FAO)

impossible (Searchinger et al. 2018). Agricultural lands have a significant impact on the earth's C and nitrogen (N) cycles due to their large size and intensive management, and agricultural activities result in releases of all three GHGs. The land use changes mainly result in the emission of CO_2, while agricultural management practices are the major contributor to N_2O (50%) and CH_4 (70%) emissions of the total anthropogenic emissions of these gases. Both are potent GHGs: N_2O has a global warming potential 296 times that of CO_2, and CH_4 has a global warming potential 23 times that of CO_2. The different sources responsible for these agricultural GHG emissions and their percent contribution are listed in Fig. 16.2 (FAO 2010).

These agricultural GHG emissions can be divided into two categories based on their production: (i) crops and (ii) livestock. The sectors are interlinked as some crops are grown for animal feed, while at the same time, the animal manure can be used as fertilizer for crops. Thereby, the allocation of the emissions to these categories is complicated and depends on accounting methodologies. Agricultural activities are the main source of the global N_2O emissions, with the share of almost 65%. For the livestock category, animal dung and urine on pastures, rangeland, and paddocks are the largest global source of N_2O emissions, accounting for 23% of the total N_2O and 4% of the total N_2O from manure management. For the crop category, synthetic N fertilizer use is the largest source, accounting for 13% of the total N_2O emissions, followed by the 11% share from decomposition of crop residues. Additionally, manure management accounts for 9% of the total N_2O emissions. Therefore, all these sources account for 74% of global N_2O emissions, with 32% share from livestock, 24% share from the crop, and 18% share from fossil fuel combustion (Fig. 16.3). Additionally, indirect N_2O emissions from agricultural activities account for another 9% of the total N_2O emissions (Fig. 16.3) (Olivier and

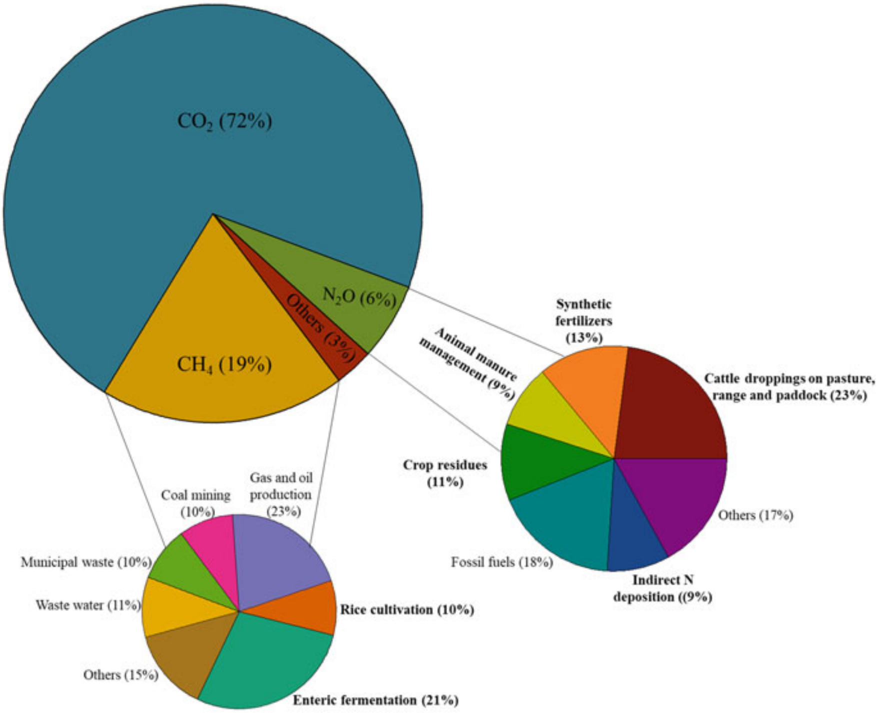

Fig. 16.3 Key drivers of nitrous oxide (N_2O) and methane (CH_4) emissions from the agricultural sector. Sections with bold letters represent agricultural sources. (Data adapted from Olivier and Peters 2020)

Peters 2020). Similarly, enteric fermentation from ruminants and rice production in flooded conditions contributes to CH_4 emissions for the livestock and crop, respectively. Cattle alone are responsible for 21% of current global CH_4 emissions, accounting for 75% of all ruminant-related CH_4 emissions (31%), followed by buffalo, sheep, and goats that have contributions of about 10%, 7%, and 5%, respectively. Rice cultivation on flooded rice fields accounts for 10% of CH_4 emissions due to the anaerobic decomposition of organic material resulting in the production of CH_4 (Fig. 16.3) (Olivier and Peters 2020). However, the CO_2 emissions are mainly derived from land use changes such as clearing of forests for agricultural development. The conversion of soil carbon (C) to CO_2 by soil microbes is accelerated in response to cultivation and growing annual crops (Verge et al. 2007). However, after few decades of soil cultivation, the soil C content is stabilized at low levels and loss as CO_2 decrease (Hutchinson et al. 2007). In addition, the use of fossil fuels for farming operations is also a source of CO_2 emissions in agriculture (Dyer and Desjardins 2003). Other sources of CO_2 emissions from agricultural lands include (a) transformations between croplands and pasture; (b) peat drainage and burning; (c) wood harvesting; (d) regrowth of forest and other natural vegetation after agricultural abandonment and harvest; and (e) soil CO_2 flux due to grassland and cropland management (Hansis et al. 2015; Houghton and Nassikas 2017; Gasser et al. 2020).

Global N_2O emissions were reported to increase to 1.1% in 2019 to a total of 2.8 $GtCO_2$-eq, similar to the annual average reported since 2014, when growth rates ranged between 0.8 and 1.3%. The different sources that were the main role players for the increase in N_2O emissions in 2019 were application of synthetic N fertilizers (+2.7%); manure deposited in pastures, rangeland, and paddocks (+1.3%); indirect N_2O from agriculture (+2.1%); and other agricultural sources (+1.1%), accounting for more than 75% of the total net increase in N_2O emissions. The countries with the largest increase in N_2O emissions in 2019 were Brazil (+2.9%), Australia (+5.9%), China (+0.9%), India (+1.6%), and the Russian Federation (+2.1%), whereas the countries with decreased N_2O emissions in 2019 were Sudan, Zaire, the Central African Republic, and the United States. Similarly, global CH_4 emissions were reported to increase at 1.3% to a total of 9.8 $GtCO_2$-eq, which was lower than the 1.8% increase in 2018. This was significantly greater than years 2015 and 2016, with an overall increase of 0.3% and 0.1%, respectively, but similar to the increase reported in years 2012, 2014, and 2017 of around 1.4%, which is also the average annual increase since 2010. Among the different sources of CH_4 emissions, livestock farming (particularly non-dairy cattle) was the second largest contributor after coal production. Among different countries that contributed most to the 1.3% growth were notably China (+2.2%) and the United States (+2.5%), with increases also seen in (in decreasing order of absolute changes) Indonesia, Brazil, the Russian Federation, Pakistan, and India. Notably, decreases were seen in Turkey, Sudan, Canada, Venezuela, Germany, and Zaire.

Global emission pathways that would limit warming to 1.5 °C or less, in line with the Paris Agreement's temperature goal, depend on significant reductions in agricultural GHGs (N_2O and CH_4) as well as net zero CO_2 emissions from fossil fuels (Leahy et al. 2020). Similarly, Wollenberg et al. (2016) also suggested a global target of reducing non-CO_2 emissions from agriculture by 1 Gt CO_2-eq below baseline by 2030 to restrict warming to about 2 °C above pre-industrial levels in 2100. The most magnificent scenarios evaluated by the IPCC (2018), which limit warming to 1.5 °C with limited or no overshoot, reduce global agricultural emissions by 16–41% (interquartile range) in 2050 compared to 2010, whereas baseline emissions increase by 24–54% over the same period. This % reduction equates to 4.8 Gt CO_2-eq in direct global agricultural non-CO_2 emissions below baseline by 2050 (Huppmann et al. 2018; Frank et al. 2019). These ambitious targets of mitigation pathways represent a large challenge, and accomplishing these targets is only possible by the implementation of effective GHG mitigation strategies from the agricultural sector.

As a major source of global emissions, the agricultural sector may also provide relatively low-cost opportunities for GHG mitigation. Agricultural GHG fluxes are complex due to interaction with other factors and variation in fluxes on spatial (varied fluxes at different places on piece of land) and temporal (variation based on time of the day) basis. However, the active management of agricultural systems offers possibilities for GHG mitigation (Smith et al. 2008). Mitigation measures in the agricultural sector include increasing C sequestration and reducing the emissions from both livestock and agricultural processes. There are two ways to achieve mitigation in the agricultural sector, i.e., through supply-side measures and demand-side measures. Supply-side measures include reducing emissions via livestock management, land management, and land use change and increasing C sequestration from afforestation. Demand-side measures include changes in eating habits and reducing food wastes; however, quantitative measures for demand-side measures are more uncertain (Smith et al. 2014). In this chapter, we will discuss mitigation strategies for GHG emissions from the agricultural sector at a global scale.

16.2 Mitigation Opportunities: Increased Sinks and Reduced Emissions

16.2.1 Increasing Carbon Sequestration

According to the recent IPCC reports, even if we can substantially reduce anthropogenic C emissions in the near future, it is necessary to make efforts to sequestering previously emitted C to ensure atmospheric C to safe levels and mitigate climate change (Smith et al. 2014). Carbon sequestration can be defined as a sustained increase in C storage (in soil or plant material or in the sea). Among these sources of C sequestration, the soil's usefulness as a C sink and drawdown solution are essential, based on global estimates of historic C stocks and projections of rising emissions (Lal 2004, 2008). Since more than one-third of the world's arable land is under agriculture (World Bank 2015) and soil C pool (2500 Gt) being 3.3 times the size of the atmospheric pool (760 Gt) and 4.5 times the size of the biotic pool (560 Gt) (Lal 2004), increasing soil C in agricultural systems will be a key component of using soils as a C sink. The C sequestration potential of global soil is estimated between 0.4 and 1.2 Gt C year^{-1} or 5–15% (1 Pg = 1 × 10^5 g) (Lal 2004). Various crop management techniques have been suggested for increasing C sequestration in soils (Janzen et al. 1998). However, large uncertainties have been reported with quantifying the impact of different crop management techniques on C sequestration and GHG mitigation. Increasing soil C sequestration could potentially remove between 0.79 and 1.54 Gt C year^{-1} from the atmosphere in a feasible manner, recognizing the large potential of soils mitigating CO_2 emissions (Laborde et al. 2021).

Due to the historical expansion of agriculture and pastoralism (Sanderman et al. 2017) and subsequent land use conversion from native ecosystems (e.g., peatlands, forests, grasslands) to arable land, 33% of the soils around the globe have been degraded and have lost much of their soil C (FAO 2019). The average amount of soil organic carbon (SOC) in the top 30 cm of native soil worldwide is about 15 Mg ha^{-1}

(Hutchinson et al. 2007). However, within the first 20 years of cultivation, about 20–30% and 50–75% of this C are lost to the atmosphere as CO_2 in temperate and tropical regions, respectively (Dumanski 2004). However, Lal (2013) reported that prolonged intensive cultivation decreases the soil C stock at the rate of 0.1–1.0% year^{-1}. The extent of C loss ranges from 10 to 30 Mg C ha^{-1}, depending on the soil type and historic land use, which is higher in soils prone to erosion, salinization, and nutrient mining than the C loss from least or undegraded soils (Lal 2013). The historical C losses from global soils are estimated to be 78 ± 12 Pg (Lal 2004; Buragohain et al. 2017). Globally, the soils of Africa are relatively low in soil organic C content with about 58% of soils containing less than 0.5% organic C and only 4% containing more than 2% organic C (Du Preez et al. 2011).

Different management practices reported to increase C sequestration include (i) reduced and zero tillage, (ii) perennial and deep-rooting crops, (iii) more efficient use of organic amendments (animal manure, sewage sludge, cereal straw, compost), (iv) improved rotations, (v) irrigation, (vi) bioenergy crops, (vii) intensification, (viii) including cover crops, and (ix) conversion of arable land to grassland or woodland (Smith 2004). The potential of these management practices for sequestering C is presented in Table 16.1. It has been estimated that implementation of appropriate management practices could help to sequester approximately 0.4–0.8 Pg C year^{-1} (Watson et al. 1996). Similarly, Lal (2010) reported that adopting suitable management practices for C sequestration at agricultural soils and restoring of degraded soils can help in sequestering about 0.6–1.2 Pg C year^{-1} for about 50 years with a cumulative sink capacity of 30–60 Pg. The potential of different management practices in sequestering C and mitigating CO_2 emissions is described below; however, prudent combination of these management practices would result in enhanced C sequestration.

Table 16.1 Carbon sequestration potential by different management practices

Management practice	Soil carbon sequestration potential (t C ha^{-1} year^{-1})
No tillage	0.38
Reduced tillage	<0.38
Set-aside	<0.38
Permanent crops	0.62
Deep-rooting crops	0.62
Animal manure application	0.38
Cereal straw application	0.69
Sewage sludge	0.26
Composting	0.38
Bioenergy crops	0.62
Organic farming	0–0.54
Extensification	0.54

All estimates are adapted from the figures in Smith et al. (2000)

16.2.1.1 Tillage Methods and Residue Management

Conventional tillage can be defined as a plow-based method which includes successive operations of plowing or turning over of soil, whereas conservation tillage is a generic term indicating at tillage methods that reduce runoff and loss of soil by erosion as compared to conventional tillage practices. Conservation tillage practices reported to increase C sequestration by reducing tillage-induced breakdown of soil aggregates resulting in the slowdown of organic matter decomposition relative to the conventional tillage and adding organic matter as residues to the surface soil (Hati et al. 2020). Different tillage practices impact both soil-aggrading and soil-degrading processes, thereby affecting soil C storage (Lal and Kimble 1997) (Fig. 16.4). Soil-aggrading processes have a positive impact on SOC and include the humification of crop residue, increase in resistant or non-labile fraction of SOC, sequestration of SOC in the formation of organo-mineral complexes, and increase in stable aggregation and deep placement of SOC in sub-soil horizons, while soil-degrading processes have a negative impact on SOC and include erosion, leaching, and mineralization. The effect of tillage on soil processes that affect C dynamic and reserves in soils can be observed in Fig. 16.4.

Several studies have reported that conservation tillage practices help in sequestering soil C in both temperate and tropical regions. Conservation tillage increased SOC by about 8% as compared to conventional tillage on an Ultisol in eastern Nigeria (Ohiri and Ezumah 1990). Several studies emphasize that conservation tillage practices have already increased soil C contents relative to levels that would have existed under conventional farming (e.g., moldboard plowing); they have estimated C sequestration rates of 0.31–0.82 Mg C ha^{-1} year^{-1} in the United States and across the world (West and Post 2002; Spargo et al. 2008; Franzluebbers 2010). However, the capability of no tillage for increasing C sequestration is still debatable. Several authors in recent years found that no-till was capable only of increasing the

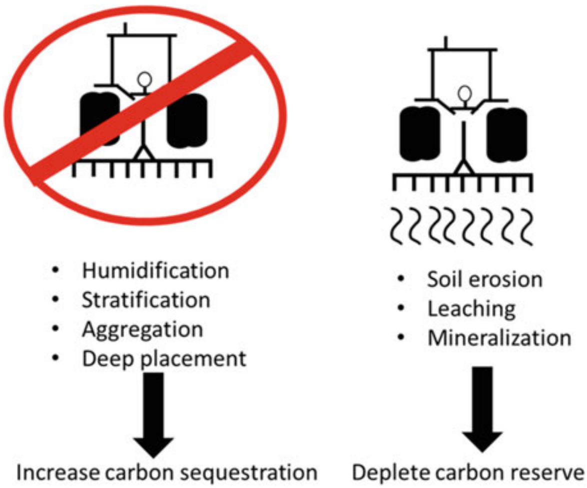

Fig. 16.4 Tillage effects on soil processes that affect C dynamics and reserves in soil

soil C in the top layer of soil, while it was compensated with the greater decrease observed in deeper layers, thereby resulting in no difference among different tillage treatments for the total C in the soil profile. However, long-term experiment results show that switching from plow-till to no-till farming is the most effective factor in crop management for SOC sequestration (Table 16.2). In a recent meta-analysis, Nicoloso and Rice (2021) found that soil C can be increased to a depth of 1 m by the intensification of no-tillage cropping systems which included double cropping, leguminous cover crops.

Crop residue management impacts the SOC dynamics as crop residues are a direct source to SOC pool. Crop residues contain approximately 45% C by dry weight (Lal 1997). Assuming that crop residues contain an average of 45% C and that approximately 15% of residue-derived C is stored as passive C in the soil, aboveground crop residues have a large potential to store SOC in the passive form on a global scale (Lal 1997). The total amount of SOC storage is determined by the quantity and quality of crop residue, plant roots, and other organic material returned to the soil, as well as the rate of their decomposition. Residue retention in combination with reduced-tillage and no-tillage practices is a viable option for increasing SOC storage in soil. In surface soil layers, under no-tillage practices, some of the residue-derived SOC gets converted into passive pool and forms organo-mineral complexes, which takes between 100 and thousands of years for decomposition. SOC accumulates when residue C inputs exceed residue C outputs and soil disturbance is kept to a minimum, while under intensive or conventional tillage practices, the decomposition of crop residues is accelerated due to good aeration, thereby resulting in reduced residue-derived C sequestration. Therefore, no-tillage practices in combination with residue retention help in the formation of the passive SOC pool and are important for long-term C sequestration.

16.2.1.2 Crop Selection and Rotation

Crop rotation refers to a planned sequence of crops grown in a regularly recurring succession on the same area, in contrast to continuous monoculture or growing a variable sequence of crops. Carbon sequestration on agricultural lands can be affected by crop rotations, climates, soils, and management practices. The use of balanced fertilization, application of organic amendments, and similarly application of crop residues in addition to intensive crop rotations can increase C sequestration levels to 5–10 Mg ha^{-1} year^{-1} since those amendments contain 10.7–18% C, which can also be helpful in the sequestration of C (Mandal et al. 2007). Different legume crops, such as peas (*Pisum sativum*), lentils (*Lens culinaris*), alfalfa (*Medicago sativa*), chickpea (*Cicer arietinum*), and sesbania (*Sesbania grandiflora*), can serve as substitute sources for N. Soil structure improvement and increased SOC content in sub-soil horizons are possible by growing deep-rooted plants. Similarly, improvement in SOC content of the sub-soil could improve in response to growing improved pastures in acid savanna soils in South America (Fisher et al. 1994). In West Africa, Lal et al. (1978, 1979) also observed significant positive effects of growing cover

Table 16.2 Impact of adopting no-tillage practices on soil carbon sequestration in different parts of the world

Location	Rotations/soils	Increase in SOC sequestration (kg ha^{-1} year^{-1})	Depth (cm)	Duration (years)	Reference
Brazil (South)	Various rotations	611	30	9	Bayer et al. (2000)
Canada	Average for groups: Gleysolic, brown, dark brown, and black (Century Model prediction)	200	–	10	Desjardins et al. (2005)
Europe	Assessment based on long-term experiments: Europe	387	25	–	Smith et al. (2000)
	United Kingdom	613	25	–	
Spain	Various rotations on Calcic Luvisol	100	30	11	López-Fando and Pardo (2001)
United States:	Various crop rotations on:				
(1) Kansas	Grundy silty clay loam	20	30	15	Havlin et al. (1990)
	Muir silt loam	62	30	15	
(2) Nebraska	Spring wheat-fallow spring	−225	30.4	12	Halvorson et al. (2002)
	Wheat-winter wheat-sunflower	542	30.4	12	
(3) Ohio	Various rotations on clay loam	566	30	30	Dick et al. (1998)
(4) Oregon	Various crops on coarse-silty mixed mesic	94	22.5	44	Rasmussen and Rhode (1988)
	Winter wheat-lentil (*Lens culinaris* Medik.)	587	20	3	Bezdicek et al. (2002)
	Winter wheat-barley with no-till management	166	20	25	
(6) Texas	Continuous corn (4y) followed by continuous cotton (4y) on sandy clay loam	15-20	20	26	Salinas-Garcia et al. (1997)
(7) Miscellaneous regions	39 paired tillage experiments	220	Various depths	5–20	Paustian et al. (1997)
World	Till to no-till 276 paired treatments excluding wheat-fallow treatments	570 ± 140	Various depths	Various time	West and Post (2002)

crops on increase in SOC content. Cover crops help in increasing soil C content only in surface layers; utilizing agroforestry (AF) systems could help in depositing C to deeper layers of soil (Meena et al. 2020; Sarto et al. 2020). The AF consists of mixture of trees, agricultural crops, and livestock to exploit the economic and ecological benefits of agroecosystem. It is a crucial leader of terrestrial C sequestration containing about 12% of the global terrestrial C (Dixon 1995). The roots of forest tress and perennial crops penetrate deeper subsurface horizons, thus placing SOC at deeper horizons far away from the range of tillage implements (Lorenz and Lal 2014). Estimating the C sequestration potential of agroforestry systems under varied ecological and management environments ranged from 0.29 to 15.21 Mg ha^{-1} $year^{-1}$ in aboveground plant biomass and 30 to 300 Mg ha^{-1} $year^{-1}$ in belowground plant parts up to a depth of 1.0 m (Nair et al. 2010). Thereby, the implementation of appropriate crop rotation and utilizing AF can help in sequestering soil C at a rate of 0.15–0.17 Mg C ha^{-1} $year^{-1}$ (Meena et al. 2020).

Bare soil is prone to erosion and nutrient leaching and contains less C than the same field under vegetation. One of the solutions for increasing C sequestration is to plant cover and catch crops that cover the soil between the main crop or in fallow periods. It is estimated that eliminating summer fallow and replacing it with some cover crop would help in sequestering soil C at a rate of approximately 0.05–0.20 Mg C ha^{-1} $year^{-1}$ (Meena et al. 2020). The basic concept of increasing C sequestration on eliminating summer fallow is that it increases soil biomass addition, resulting in increased C deposition. Also, if the soil is left bare (fallow), it is more prone to erosion by wind or water, and as most of the C is deposited in surface layers in croplands, it is more prone to wind and water erosion and decomposition. Soil erosion alone is responsible for the loss of 1.1 Pg C $year^{-1}$ (Meena et al. 2020). Legumes enhance biological diversity, increase N input (via N fixation), and improve crop residue quality and overall soil C flux (Lal 2004). The greater the biodiversity of an ecosystem, the more will be the sequestration capacity. The unique advantage of cover crops over the other management options is that they not only enhance the SOC stock but also reduce the C loss, unlike organic manures. Hence, replacing the fallow period with cover cropping improves the soil quality by enriching SOC through their biomass and promoting soil aggregation and protecting the surface soil from runoff and erosion.

16.2.2 Reducing Nitrous Oxide Emissions

In recent years, there has been a growing interest in the possibility of mitigating climate change by reducing emissions of non-CO_2 GHGs. Agriculture is the largest anthropogenic source of N_2O, one of the most important non-CO_2 GHGs because it is a long-lived GHG (about 114 years) and a major source of NO in the stratosphere (Reay et al. 2012). For the past few decades, the amount of N_2O in the atmosphere has increased almost linearly at approximately 0.7 ppb or 0.26% $year^{-1}$ (Smith 2010). The IPCC (2001) reported that the increased microbial production of N_2O in expanding and fertilized agricultural lands is the main driver of this increase. With a growing human population and the resulting need for more food production, agricultural land area and N_2O emissions are expected to increase in the coming decades. We assume that changes in N cycling in soil systems have influenced increases in atmospheric N_2O over the past century and will help dictate future changes since roughly 70% of the N_2O emitted is derived from soils (Bouwman 1990; Braker and Conrad 2011). Among different continents, Asia is the continent with the largest N_2O emissions, reflecting its large population and agricultural area (Oenema et al. 2014). On a per capita basis, Asia has the lowest estimated N_2O emissions, followed by Africa and Europe. Expressed per surface area of agricultural land, emissions are highest in Asia and Europe and least in Oceania and Africa. The largest source of N_2O emissions in Asia, Europe, and North America is fertilizer N, while manure N from grazing animals is the largest source in Africa, Latin America, and Oceania. Therefore, the main source for N_2O emissions from the agricultural land includes lower efficiency of synthetic N fertilizers applied to croplands and urine and dung excreted by the animals, either in pastures or in confinements (stables, barns, sheds, corrals). In general, management practices that optimize the natural ability of the crop to compete with processes where plant available N is lost from the soil-plant system (i.e., NH_3 volatilization, denitrification, and leaching) and directly lowering the rate and duration of the loss processes can reduce N_2O emissions from synthetic N fertilizers and organic N sources such as crop residue and animal excreta (Doerge et al. 1991). In this section, we have described different management strategies which have the potential for mitigating N_2O emissions from croplands and grazing lands around the world.

16.2.2.1 4R of Fertilizer Management

The major source of N_2O emissions from croplands is the application of N fertilizers. In addition, increasing demands for food around the world would not allow reductions in the usage of N fertilizers to decrease N_2O emissions. Moreover, crop improvement in major crops such as corn (*Zea mays* L.) increases the dependency on N fertilization as yields increase over time (Ciampitti and Vyn 2012). Therefore, the only solution to reduce N_2O emissions from croplands without jeopardizing global food production is to enhance nitrogen use efficiency (NUE) (Ciampitti and Vyn 2014; Singh et al. 2019). The uptake of N fertilizer by crops varies widely across the world, and global cereal NUE is reported to be only 33% (Raun and Johnson 1999). Additionally, the insignificant trend of increase in global cereal NUE from 2002 to 2015 reported by Omara et al. (2019) is a cause of concern. It is estimated that each year, approximately 1.5 Tg of N is lost as N_2O to the atmosphere because of the application of synthetic N fertilizers to agricultural ecosystems (Mosier et al. 1996). This accounted for about 44% of the anthropogenic input and 13% of the total annual N_2O input into the atmosphere. However, the contribution of synthetic N fertilizers to N_2O emissions is still thought to be underestimated. Additionally, N_2O production from other major N sources such as animal manures and biological N fixation has not been included in the abovementioned estimates. To meet the needs of rapidly expanding population, the use of N fertilizers is also projected to increase in the coming years for increasing global food production. Thereby, it is very important to reduce the loss of N fertilizers as N_2O emissions and increase the N use efficiency. This will result in mitigating GHG emissions from different N fertilizers and will be economically beneficial for the producers. The "4R" approach of using the right source, right rate, right timing, and right placement is an accepted framework for reducing loss of N fertilizers as N_2O and increasing crop N use efficiency. Modifying just one of the 4R components may not be enough to reduce N_2O emissions (Decock 2014). Different studies demonstrated that the use of right time alone (delayed and/or split application) (Phillips et al. 2009; Zebarth et al. 2012) or right source (e.g., urea-containing microbial inhibitors) (Parkin and Hatfield 2013; Sistani et al. 2011) has been not very successful in mitigating N_2O emissions. The 4R technique is effective when you have site-, soil-, and crop-specific knowledge and information, accompanied with appropriate technologies and best management practices. It has been reported that implementation of 4R strategy could help in achieving N uptake more than 70% for many cereals (Snyder and Fixen 2012).

While choosing the best fertilizer source may appear to be a simple task, there are several factors that ultimately influence this decision. Selecting an appropriate fertilizer source starts with an assessment of which nutrients are necessary, and this information comes from some form of site diagnostics such as soil testing. The responses of different N fertilizers (nitrate-, ammonium-, or urea-based) to N_2O emissions are very dynamic depending on soil conditions (well-drained or moist conditions), air temperatures, and other climatic conditions. Therefore, there is possibility of decreasing N_2O emissions from N fertilizers and increasing N use efficiency by choosing specific fertilizers for a particular location. Another option for choosing the right source of N fertilizer is the use of "enhanced efficiency fertilizers" instead of conventional fertilizers. Enhanced efficiency fertilizers have been reported to improve N fertilizer use efficiency by increasing the availability of N to crops while reducing N loss to the environment (Snyder 2017; Zhang et al. 2015) including N_2O emissions (Akiyama et al. 2010; Ju et al. 2011). Experiments have shown that these types of fertilizer can decrease N_2O emissions by 35–38% relative to conventional N fertilizer (Akiyama et al. 2010). Bastos et al. (2021) and Arango and Rice (2021) found a 66% reduction in N_2O emissions with a combination of placement and a nitrification inhibitor.

Nitrous oxide emission from N fertilizer application can be reduced by synchronizing with plant N demand. The N uptake during the beginning of the growing season of the crop is lower, increases exponentially during vegetative growth, and drops sharply at crop maturity. Therefore, applying N fertilizer a few weeks after planting rather than at or before planting increases the likelihood that the N will end up in the crop rather than be lost to the atmosphere as N_2O emissions. Soil moisture is the major driver of the N_2O emissions from soil as it regulates the availability of oxygen to microbes. Impacted by different soil types, the maximum N_2O emissions are emitted when soil water-filled pore space ranges from 60 to 90% (Wang et al.

2021; Bastos et al. 2021). Therefore, application of N fertilizer during high soil moisture levels may also help in reducing N_2O emissions. Split N applications to crops result in reduced concentrations of soil mineral N in the early growth stage of crops. Application of the second portion of N during the active growth phase, when N uptake is at maximum, also reduces the potential for N_2O emissions to occur (Van Groenigen et al. 2010). Split application of N was reported as an effective strategy to reduce N_2O emissions from potato cultivation (Burton et al. 2008). In corn production, a single application of N was reported to emit 35% more N_2O compared to split applications (Fernández et al. 2016).

In addition to the right timing, applying N more than the crop requirement increases soil ammonium and nitrate concentrations in soils (Andraski et al. 2000). As a consequence, relatively higher N_2O emissions can occur when compared with applications at the required rate (McSwiney and Robertson 2005; Ma et al. 2010). To know the amount of N fertilizer application, the proper information about the site soil and crop need is required. Stehfest and Bouwman (2006) also reported the rate of N fertilizer application to be the strongest predictor of N_2O emissions in their extensive review of published articles all over the world. Although the reported mean N_2O emission factor is 1.2%, which means for every 100 kg of N input, 1.2 kg of N is lost as N_2O emissions (Albanito et al. 2017), results from a growing number of field experiments indicate that the fraction of applied N emitted as direct N_2O increases with increasing rate of N application (McSwiney and Robertson 2005; Ma et al. 2010; Hoben et al. 2011; Shcherbak et al. 2014; Millar et al. 2018). Therefore, using the single emission factor across the fertilizer rates may result in an underestimation of fertilizer-induced N_2O emissions when fertilizer addition exceeds crop demand.

Right placement of N fertilizer in the soil also helps to reduce N_2O emissions. For example, the application of urea in a narrow band close to plant roots instead of its application by broadcast helps to reduce N_2O emissions. Also, different crops have exhibited different root growing habits and require specific N fertilizer placement method for the enhancement of N use efficiency. For corn, shallow instead of deep placement of N fertilizers is reported to decrease N_2O emissions and increase N use efficiency (Breitenbeck and Bremner 1986). The precision fertilizer application tools are also reported to help reduce N_2O emissions and increase N use efficiency. This is because precision fertilizer application helps to access the spatial variability in the field, recommending less N fertilizer application in areas of the field with low yield potential, thereby helping to avoid N fertilizer wastage on locations in the field that are not likely to respond to N fertilizer application. Precision fertilizer application reduced the average N fertilizer rate by 25 kg N ha^{-1} in one study, resulting in significant reductions in N_2O emissions (Sehy et al. 2003).

16.2.2.2 Grazing and Manure Management

The relative importance of microbial processes that lead to N_2O emissions from animal manures will be determined by the manure environment, which is influenced by local management practices and climate, both of which vary between regions. A large portion of N_2O emissions resulting from manure are produced in manure-amended soils by microbial nitrification under aerobic conditions and partial denitrification under anaerobic conditions, with denitrification producing more N_2O (Hockstad and Hanel 2018). This manure can be deposited by the grazing animals in grassland-based systems or applied manually after collection and storage from confined-animal feeding systems. Under continuous stocking, specific hotspots of mineral N, or higher overall amounts of mineral N, are expected to appear in soils within grazed paddocks or portions of grazed paddocks. This premise is based on the fact that cattle have more opportunity (more time) to congregate in local areas (e.g., water sources, near to borders, shady areas) of paddocks, resulting in less-even N distributions (Singh et al. 2019). It is reported that animals spend 27% of their time and deposit around 49% of all N in consumed biomass to these areas (Augustine et al. 2013). Additionally, N_2O emissions from the pen surfaces of open-lot dairy or beef feedlot facilities can also be significant due to improper handling and storage of the manure (Montes et al. 2013).

For grassland-based systems, changing the form of grazing management and intensity of grazing pressure are among the strategies available to reduce N_2O emissions. Due to the effects on soil compaction and other physical, chemical, and biological properties of soils, higher stocking rates applied to pastures result in higher N_2O emissions from grazing lands. Also, stocking at high rates may result in the consumption of more low-quality forage by animals, which has an impact on both animal performance and greater N_2O emissions (Wang et al. 2015). Thus, the management of stocking density (animal numbers ha^{-1} year^{-1}) applied to graze paddocks is an essential practice for mitigating N_2O emissions. Increased N_2O emissions due to increased deposition of manure and urine could be caused by intensive forms of stocking. Further, the anaerobic conditions caused by increased soil compaction in grazing paddocks help to support N_2O emissions from these deposits. Reduced dietary N and increased mineral content of biomass available for grazing are two other ways to reduce N_2O emissions from grazing lands. N excretion in urine is reduced when dietary N is reduced. Additionally, inhibiting nitrification from N hotspots in grazing lands could be a useful strategy for reducing N_2O emissions. Approximately 55% of the total daily N_2O emissions from grazing paddocks is contributed by N hotspots which include urine patches, dung pats, shaded areas, and areas near water troughs (Cowan et al. 2015). The primary source of significant emissions from these hotspots is cow urine and dung, which enriches the soil with nutrients, particularly N, and moisture, creating ideal conditions for N_2O emissions. Different mitigation strategies for reducing N_2O emissions from these areas have been recommended, including restricted grazing during wet periods that favor denitrification, feeding cattle low-N diets, using stand-off pads, application of soil amendments (i.e., lime) to increase soil pH to shift the balance between N_2O and non-greenhouse N_2, or use of zeolite to capture soil NH_4. The blanket application of nitrification inhibitors like dicyandiamide in combination with urease inhibitors like nBTPT has been recommended as the best approach to reduce N losses from grazing lands among all the abovementioned strategies (Zaman and Nguyen 2012). However, there is a need of research for investigating timing, type, rate, and cost associated with nitrification inhibitor application in different regions for mitigating N_2O emissions from grazing lands.

In confined-animal feeding systems, manure is typically collected and must be managed from the point of excretion through storage, treatment, and finally applying to land. To reduce the N_2O emissions from animal manures during its storage, it is suggested that solid manures need to be kept covered. However, there are some studies with contradicting results reporting increased N_2O emissions of manure covering (Table 16.3) (Petersen et al. 2013). Additionally, the application of nitrification inhibitors to the manures while storage has the potential to reduce N_2O emissions (Petersen 2018). According to one meta-analysis, the reduction in N_2O emissions due to nitrification inhibitor application to stored manures can range from 40 to 50% (Qiao et al. 2015). Likewise for N fertilizer application, different factors such as method, rate, placement, and timing of application according to crop nutrient requirements are crucial for mitigating N_2O emissions from manures.

16.2.3 Reducing Methane Emissions

Methane is a GHG currently contributing to about 15 % of global anthropogenic GHGs emitted every year when assuming a greenhouse warming potential of 25 times CO_2 over 100 year and 50.6% of anthropogenic CH_4 emissions are released as a result of agricultural activities. China followed by India, Brazil, the United States, Indonesia, Australia, Russia, Argentina, Thailand, and Nigeria are ten major contributors of the CH_4 emissions from the agricultural sector, constituting about 54.6% of the global emissions. Among different agricultural activities, 59.8% of CH_4 emissions are contributed by the enteric fermentation followed by emissions from rice cultivation, other agricultural activities, and manure management (Karakurt et al. 2012). Enteric fermentation refers to the process of foods being fermented by microbes in an animal's digestive system. As a byproduct of this process, CH_4 is released by animals exhaling (Karakurt et al. 2012). The majority of CH_4 emissions in this sector is contributed by domesticated ruminants like cattle, buffalo, sheep, goats, and camels. However, other domesticated non-ruminants such as swine and horses also contribute to CH_4 emissions through enteric fermentation, but emissions per animal species vary significantly. Another major contributor to CH_4 emissions from the agricultural sector includes rice cultivation which contributes approximately 11% of global anthropogenic CH_4 emissions (IPCC 2013). In a flooded rice field, the decomposition of organic materials in an environment without oxygen results in the release of CH_4. The breakdown of organic components under flooded rice conditions consumes available oxygen in soil and water rapidly, and methanogenic bacteria produce CH_4 when the oxygen in the environment is depleted. Additionally, manure storage from confined-animal feeding systems in liquid form can contribute to CH_4 emissions. Storing manures in liquid systems such as lagoons, ponds, or pits results in anaerobic conditions, resulting in CH_4 emissions (Steed and Hashimoto 1994). However, the amount of CH_4 from manure varies with

Table 16.3 Effects of different management options on CH_4, N_2O, and combined CH_4+ N_2O emissions from manure storage

Type of storage	Management option	Nitrous oxide	Methane	$N_2O + CH_4$	References
Solid manure	Forced v. passive composting	−35	−90	−78	Amon et al. (2001)
		−41	+32	−7	Amon et al. (2001)
		+44	−81	−34	Pattey et al. (2005)
			−28		Hao et al. (2001)
	Straw cover	−42	−45	−42	Yamulki (2006)
		−11	−50	−14	Yamulki (2006)
	Plastic cover	−70	−6	−36	Chadwick (2005)
		+2000	−81	−17	Chadwick (2005)
		−54	+120	+111	Chadwick (2005)
		−99	−87	−98	Hansen et al. (2006)
		−32			Thorman et al. (2006)
		+304			Thorman et al. (2006)
Liquid manure	Straw cover	+57	−25		VanderZaag et al. (2009)
		+100	−27	−23	VanderZaag et al. (2009)
			+37	−24	Guarino et al. (2006)
			+3		Guarino et al. (2006)
			+7		Guarino et al. (2006)
			−28		Guarino et al. (2006)
	Solid cover	+432	+22	+238	Berg et al. (2006)
		+30	−32	+1	Amon et al. (2007)
		−4	−70	−52	Amon et al. (2007)
		−50	−37	−48	Amon et al. (2007)
		−13	−14	−13	Clemens et al. (2006)
		+20	−16	−11	Clemens et al. (2006)
		+2	−29	−4	Clemens et al. (2006)
		−19	−14	−16	Clemens et al. (2006)

"+" represents higher emissions (%) and "–" lower emissions (%) compared with the reference (untreated) manure. The comparison of systems is based on CO_2 equivalents. Data is adapted from Peterson et al. (2013)

respect to the storage type, ambient temperature for storage, and manure composition. Open biomass burning, savanna burning, agricultural residue burning, and open forest clearing burning are other agricultural sources of CH_4 emissions. In this section, we will be discussing strategies to mitigate CH_4 emissions from different agricultural sources.

16.2.3.1 Improving Rumen Fermentation Efficiency and Productivity of Animals

Due to their unique digestive system, which includes a rumen, ruminant animals such as cattle, buffalo, sheep, and goats produce a lot of CH_4. The methanogenic archaebacterium responsible for CH_4 production is located mainly in the rumen, and its growth is affected by diet and other nutritionally related characteristics such as level of intake, feeding strategies, quality of fodder, and fodder concentrate ratios (Karakurt et al. 2012). Therefore, numerous nutritional technologies have been evaluated to increase rumen fermentation efficiency and reduce CH_4 production, such as direct inhibitors, feed additives, propionate enhancers, CH_4 oxidizers, probiotics, defaunation, diet manipulation, and hormones. Up to 40% reduction in CH_4 emissions is reported as a result of dietary manipulation depending on the degree of change and nature of the intervention (Benchaar et al. 2001). Dietary manipulation includes improving forage quality or changing the proportion of diet and dietary supplementation of feed additives that directly either inhibit methanogens or alter the metabolic pathways leading to a reduction of the substrate for methanogenesis. Forage quality can be improved by providing high-quality forage as it contains higher amounts of easily fermentable carbohydrates and less neutral detergent fiber, leading to a higher digestibility and passage rate, thereby resulting in lower CH_4 production, while more mature forage has a higher C:N ratio, which results in decreased digestibility and higher CH_4 production (Beever et al. 1986). Feeding legume forage results in lower CH_4 emissions as it contains condensed tannins, a low fiber content, high dry matter intake, and fast passage rate (Beauchemin et al. 2008). In general, feeding C_3 plant yields less CH_4 emissions than that from C_4 plants (Archimède et al. 2011). Similarly, replacing grass silage with maize silage helps in reducing CH_4 emissions from enteric fermentation. The reason is the same that grass silage is usually harvested at a later stage of maturity and contains lower content of digestible organic matter, lower sugar, and N contents, whereas maize silage provides higher contents of dry matter with readily digestible carbohydrates, e.g., starch, increasing the dry matter intake and animal performance (Beauchemin et al. 2008). Additionally, concentrates, fat supplementation, organic acids, essential oils, ionophores, and probiotics as feed additives reduce CH_4 emissions from enteric fermentation. Another method suggested for increasing rumen fermentation is the possibility of breeding animals with low CH_4 emissions. However, Eckard et al. (2010) suggested that breeding for reduced CH_4 production is unlikely compatible with other breeding objectives. Another way to reduce enteric CH_4 emissions is to increase the milk yield of dairy animals. However, increasing productivity will only reduce the total enteric CH_4 emissions if the amount of milk produced is kept constant by reducing the number of animals (Sirohi et al. 2007). Diet not only has a direct impact on CH_4 emissions from intestinal fermentation, but it also has an indirect impact on CH_4 emissions during storage by influencing manure composition (Hindrichsen et al. 2005).

16.2.3.2 Manure Management

Methane production is significantly decreased under dry and aerobic conditions; thereby, switching from liquid to dry manure management systems would help minimize CH_4 emissions from manure storage and handling. Methanogenesis is dependent on temperature, being lower under cooler temperatures. Therefore, storing slurry at cooler temperatures (10 °C) could result in 30% to 46% reduction in CH_4 emissions (Table 16.4). In cold and temperate climates, the temperature difference between animal housing and outside manure storage is significant. Therefore, by frequent removal of manure from housing to outside storage could help mitigate CH_4 from manure (Table 16.5). While storage, aeration of the solid manure left for composting also helps reduce CH_4 emissions from manure as it helps maintain aerobic conditions. Similar to N_2O emissions, covering both liquid and solid manures using straw or plastic sheets is also a mitigation strategy for CH_4 emissions from manure. However, some studies also reported contradicting results showing increased CH_4 emissions on manure covering (Chadwick 2005; Berg et al. 2006). Another method reported to mitigate CH_4 emissions from manure is its separation, herein defined as a process whereby a fraction of slurry particles is isolated by one of the several mechanical separation processes. Separate storage of the liquid and solid fractions after manure separation has, in most cases, but not always, resulted in lower CH_4 emissions (Table 16.4). Anaerobic digestion of manure is another strategy for mitigating CH_4 emissions where methanogenesis is optimized for breaking down degradable organic matter in manure and transforming it into biogas. As CH_4 is collected and used as fossil fuel, it reduces CH_4 emissions during storage. The potential of anaerobic digestion for reducing CH_4 emissions from manure reported under different studies can be found in Table 16.4. Additionally, treatment of slurry/manure using sulfuric acid is reported to reduce CH_4 emissions by 67% to 99% during 3-month storage period (Table 16.4). Manure aeration is an efficient way for mitigating CH_4 emissions because aerobic conditions are maintained. Amon et al. (2006) reported a reduction in CH_4 emissions (by 57%), with aeration of cattle slurry, while Martinez et al. (2003) reported reductions in CH_4 emissions of 70% to 99% after aeration of pig slurry. Therefore, using these mitigation strategies alone or in combination with others could help reduce CH_4 emissions during manure storage and handling.

Table 16.4 Effects of different management options on CH_4, N_2O, and combined CH_4 + N_2O emissions from manure treatment

Management option	Type of manure	Nitrous oxide	Methane	$CH_4 + N_2O$	References
Manure separation	Pig slurry (5 °C)	0	−8	−8	Dinuccio et al. (2008)
	Pig slurry (25 °C)		+3	+41	Dinuccio et al. (2008)
	Cattle slurry (5 °C)	0	+4	+4	Dinuccio et al. (2008)
	Cattle slurry (25 °C)	0	−9	−9	Dinuccio et al. (2008)
	Cattle slurry	+1133	−34	−23	Fangueiro et al. (2008)
	Cattle slurry + wooden lid	+10	−42	−39	Amon et al. (2006)
	Pig slurry		−93	−29	López-Mosquera et al. (2011)
	Cattle slurry		−42	+25	López-Mosquera et al. (2011)
	Pig slurry		−18		Martinez et al. (2003)
	Cattle slurry		−40		Martinez et al. (2003)
Anaerobic digestion	Cattle slurry	−9	−32	−14	Clemens et al. (2006)
	Cattle slurry	+49	−68	−48	Clemens et al. (2006)
	Cattle slurry + wooden lid	+41	−67	−59	Amon et al. (2006)
Aeration	Cattle slurry	+144	−57	−43	Amon et al. (2006)
	Pig slurry		−99		Martinez et al. (2003)
	Pig slurry		−70		Martinez et al. (2003)
Dilution	Pig slurry		−35		Martinez et al. (2003)
	Cattle slurry		−57		Martinez et al. (2003)
Additives					
NX_{23}	Pig slurry		−47		Martinez et al. (2003)
Stalosan	Pig slurry		−54		Martinez et al. (2003)
Biosuper	Pig slurry		−64		Martinez et al. (2003)
Sulfuric acid (pH 6)	Cattle slurry		−87		Petersen et al. (2012)
	Pig slurry		−99		Petersen et al. (2012)
	Pig slurry		−94		Petersen et al. (2012)

"+" represents higher emissions (%) and "–" lower emissions (%) compared with the reference (untreated) manure. The comparison of systems is based on CO_2 equivalents

16.2.3.3 Reducing CH_4 Emissions from Flooded Rice Cultivation

Rice is grown on over 140 million hectares around the world and is the world's most widely consumed staple food. About 90% of the world's rice is produced and consumed in Asia, and 90% of rice land is flooded, at least temporarily (Wassmann et al. 2009). During the growing season, the soil redox potential decreases significantly due to flooded and anaerobic conditions, creating an environment conducive to methanogenesis, thereby resulting in CH_4 emissions. Estimates of global CH_4 emissions from paddy soils range from 31 to 112 Tg year^{-1}, accounting for up to 19% of the total emissions, while 11% of global agricultural N_2O emissions come from rice fields (US-EPA 2006; IPCC 2007). Rice production may need to increase to keep pace with the growing demand; efficient and sustainable management is needed to mitigate CH_4 emissions from rice paddy fields while maintaining high rice yields. Water regime and organic inputs determine most CH_4 emissions from rice fields, but soil type, weather, tillage management, residues, fertilizers, and rice cultivar also play a role. Therefore, changing the water management with soil submergence to a limited period seems to be the most promising option for mitigating CH_4 emissions from flooded rice fields. Midseason drainage (a common irrigation practice adopted in major rice-growing regions of China and Japan) and intermittent irrigation (common in northwest India) reduce CH_4 emissions by over 40%. Under midseason drainage, the time under anaerobic conditions is reduced, and most of the CH_4 in the soil is oxidized when exposed to air, which raises the soil redox potential to levels that prevent methanogenesis (Souza et al. 2021). However, the field needs to be reflooded before the soil moisture level falls a critical plant water stress level and prevents yield loss. Also, practicing early-season drainage in combination with midseason drainage is reported to be more effective than only midseason drainage as it helps reduce about 80–90% of CH_4 emissions. As the main solution for reducing CH_4 emissions for flooded rice is to limit soil submergence to a limited period, switching flooded rice cultivation to upland rice cultivation also reduces CH_4 emissions. However, the adoption of upland rice cultivation is not preferred because its production potential is much lower (Neue 1993). Another option for minimizing CH_4 emissions from flooded rice is by the adoption of direct seeding instead of transplanting. However, there are debates about the profitability of direct seeds rice due to the weed problem.

In addition to water management, fertilization management is relevant for mitigating CH_4 emissions from rice cultivation. Soil fertilization using fresh organic matter amendments, such as rice straw and green manures, significantly increases CH_4 production and emissions. Therefore, organic amendments may need to be minimized to reduce CH_4 emissions from wetland rice fields. However, sometimes, use of green manures and crop residues is the only source of soil nutrition for resource-limited farmers. In general, due to the availability of chemical fertilizers and responsive rice cultivars, organic amendments have declined in recent years. Among different chemical fertilizers, sulfate-containing fertilizers mitigate CH_4 emissions (Ro et al. 2011). This is because sulfate-reducing bacteria compete with methanogens for limited hydrogen. Use of urea-encapsulated calcium carbide as an N fertilizer if flooded rice is reported to mitigate CH_4 emissions due to slow release of acetylene (Bronson and Mosier 1991).

16.2.4 Quantifying and Modeling GHG Fluxes

The improvement in accuracy and robustness of the estimates of the GHG implications of the abovementioned practices is necessary as the agricultural sector plays an important role in addressing climate change. Particularly, the capacity to estimate

Table 16.5 Effects of different management options on CH_4, N_2O, and combined CH_4 + N_2O emissions from animal housing

Management option	Animal category	Nitrous oxide	Methane	N_2O + CH_4	References
Straw bedding	Fatteners	+106	−2	+29	Philippe et al. (2007)
	Gestating sows	+383	−9	+131	Philippe et al. (2007)
	Weaned pigs		−18	+22	Cabaraux et al. (2009)
	Dairy cattle	+85	+33	+48	Edouard et al. (2012)
Sawdust v. straw	Weaned pigs	+286	−51	+195	Nicks et al. (2003)
	Fatteners	+6867	−33	+286	Nicks et al. (2004)
	Fatteners	+7600	+100	+667	Kaiser (1999)
Wood shavings v. straw	Laying hens	+259	+319	+275	Mennicken (1998)
Cooling	Pigs		−31		Sommer et al. (2004)
	Fatteners		−43		Groenestein et al. (2012)
	Nursing sows		−46		Groenestein et al. (2012)
	Gestating sows		−33		Groenestein et al. (2012)
	Weaned pigs		−30		Groenestein et al. (2012)
Frequent manure removal	Pigs	−39	−56	−51	Amon et al. (2007)
	Pigs		−40		Haeussermann et al. (2006)
	Weaned pigs	0	−50	−50	Groenestein et al. (2012)
	Fatteners	0	−86	−86	Groenestein et al. (2012)

"+" represents higher emissions (%) and "–" lower emissions (%) compared with the reference (untreated) manure. The comparison of systems is based on CO_2 equivalents

CH_4 and N_2O emissions and changes in emissions needs to be strengthened, and a global monitoring system to provide measurements of soil C stocks over time should be established. Making informed decisions about the most appropriate mitigation strategies requires a thorough understanding of how much C can be sequestered or how various practices can reduce much GHG emissions. However, significant gaps remain, particularly in developing countries, where there are still many questions about the sources of agricultural emissions, as well as a lack of methods and methodologies for monitoring emissions through supply chains and evaluating the GHG impacts of investments. Additionally, the mathematical models can articulate the factors that control GHG fluxes and soil C stock changes. Therefore, a combination of field measurements and models considering farming systems is the most effective method for estimating global-scale agricultural emissions and sinks, as well as forecasting changes in emissions due to changes in management practices, environmental and economic conditions, or government policies (Table 16.5).

The rate of GHG emissions from soils and/or uptake can be measured directly using the chamber method and micrometeorological techniques. However, because emission rates are highly variable in both space and time, measuring flows of these gases over areas and time periods of interest poses significant challenges. For example, following a rainstorm or fertilization, N_2O emission rates can change 100-fold or more (Smith et al. 2000), and similar changes in CO_2 emission rates occur after tillage (Reicosky et al. 1997). Therefore, calculating annual flux rates demands frequent sampling to adequately represent large, short-term fluxes and avoid under- or over-estimation of fluxes. Due to the high spatial variability of flux rates, either several small areas within a field must be sampled and averaged, or the measurement technique must integrate fluxes over a relatively large area. In addition, automated chamber systems can be utilized for overcoming the error due to temporal variability.

Mathematical models can be used for articulating the factors that control GHG fluxes and soil C stock changes. There are two basic types of models: (i) empirical and (ii) "process-oriented" models. Empirical models use field measurements to determine statistical relationships between soil C stocks and environmental and management factors (e.g., IPCC 1997; Ogle et al. 2003), whereas more dynamic, "process-oriented" models attempt to simulate the biological, chemical, and physical processes that control GHG dynamics. Process-oriented models are useful to portray the effect of combinations of management practices as well as soil and climate conditions. Several dynamic, process-based models have been developed to simulate soil C stock changes and N_2O and CH_4 fluxes from soil.

16.3 Conclusions

Global agriculture has significant potential to reduce GHG emissions and sequester C in soils using currently available technology. However, because there are so many variables that influence emission and sequestration processes, some practices that reduce one gas emissions may increase emissions of another. Promoting practices that maintain or increase C stocks while also increasing the efficiency of agricultural inputs (e.g., fertilizer, irrigation, pesticides, animal feed, and animal waste) is the key to reducing net GHG emissions from agriculture. To achieve the best overall mitigation results, GHG mitigation practices should address both C stocks and N_2O and CH_4 emissions. The largest potentials for soil C sequestration are associated with adoption of no-till practices, reduced fallow, use of cover crops, and conservation set-asides with perennial grasses and trees on highly erodible cropland. Nitrous oxide emissions from soils constitute the single largest agricultural GHG source. More efficient use of N fertilizer and manure, increasing the overall efficiency of N use for crop improvement, as well as additives that inhibit the formation of N_2O in soils could help in the reduction of N_2O emissions. Methane emissions are mainly contributed through enteric fermentation and emissions from stored manure from livestock production or from flooded rice cultivation. Manure management systems that capture and combust CH_4 can provide a renewable energy source that both is helpful in reducing CH_4 emissions and can displace fossil fuels. Improved production technologies (e.g., improved feed quality, CH_4-suppressing feed additives, and animal breeding) can reduce enteric CH_4 emissions, increase livestock production, and perhaps improve profitability. For rice cultivation, avoiding the use of organic inputs, fertilizer management, using nitrification inhibitors and irrigation management techniques such as midseason drainage or intermittent drainage can help in mitigating CH_4 emissions.

With respect to quantification, this report finds that direct field measurement is viable, although at times expensive, for assessing C sequestration; field measurement of CH_4 and N_2O is not yet ready for wide implementation. Direct measurement appears best suited for programs focused on innovative new practices for which research is lacking. In contrast, modeling will likely be most efficient for scaling up known management practices well supported by research and modeling capacity. Important data gaps remain for program or project implementation particularly management data for establishing baseline conditions. Additional work is needed to assess potential reversal rates for the subset of management practices for which this could be a problem.

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