

# Chapter 8

## Climate-Smart Agriculture Practices for Mitigating Greenhouse Gas Emissions



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**Abstract** Agricultural lands make up approximately 37% of the global land surface, and agriculture is a significant source of greenhouse gas (GHG) emissions, including carbon dioxide (CO<sub>2</sub>), methane (CH<sub>4</sub>) and nitrous oxide (N<sub>2</sub>O). Those GHGs are

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responsible for the majority of the anthropogenic global warming effect. Agricultural GHG emissions are associated with agricultural soil management (e.g. tillage), use of both synthetic and organic fertilisers, livestock management, burning of fossil fuel for agricultural operations, and burning of agricultural residues and land use change. When natural ecosystems such as grasslands are converted to agricultural production, 20–40% of the soil organic carbon (SOC) is lost over time, following cultivation. We thus need to develop management practices that can maintain or even increase SOC storage in and reduce GHG emissions from agricultural ecosystems. We need to design systematic approaches and agricultural strategies that can ensure sustainable food production under predicted climate change scenarios, approaches that are being called climate-smart agriculture (CSA). Climate-smart agricultural management practices, including conservation tillage, use of cover crops and biochar application to agricultural fields, and strategic application of synthetic and organic fertilisers have been considered a way to reduce GHG emission from agriculture. Agricultural management practices can be improved to decreasing disturbance to the soil by decreasing the frequency and extent of cultivation as a way to minimise soil C loss and/or to increase soil C storage. Fertiliser nitrogen (N) use efficiency can be improved to reduce fertilizer N application and N loss. Management measures can also be taken to minimise agricultural biomass burning. This chapter reviews the current literature on CSA practices that are available to reduce GHG emissions and

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increase soil C sequestration and develops a guideline on best management practices to reduce GHG emissions, increase C sequestration, and enhance crop productivity in agricultural production systems.

**Keywords** Agriculture · Carbon dioxide · Climate-smart agriculture · C sequestration · GHG · Methane · Mitigation · Nitrous oxide

## 8.1 Introduction on Climate-Smart Agriculture Practices and Greenhouse Gas Emissions

Agriculture is a major source of greenhouse gases (GHGs) that affect climate change and is also itself a victim of climate change. Agricultural lands make up 37.6% of the global land surface, and agriculture is a significant source of GHG emissions (IPCC 2014; Smith et al. 2008), where CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O are the major forms of trace gases that are responsible for the majority of the global warming effect. Agricultural GHG emissions are associated with agricultural soil management (e.g. tillage), use of both synthetic and organic fertilisers, livestock management, burning of fossil fuel for agricultural operations and burning of agricultural residues. In particular, agriculture can be the source for 52% and 84%

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of global anthropogenic emissions of CH<sub>4</sub> and N<sub>2</sub>O, respectively (Smith et al. 2008). Since the global warming potentials of CH<sub>4</sub> and N<sub>2</sub>O are much higher than that of CO<sub>2</sub> based on per unit mass and a 100-year time frame (IPCC 2014), advanced concepts are required to reduce agricultural emissions of CH<sub>4</sub> and N<sub>2</sub>O.

In addition to causing increased levels of GHG emission, human settlement in previously unpopulated areas means that natural ecosystems are converted to agricultural production, with 20–40% of the SOC lost following cultivation and with most of that loss occurring in the first a few years (Davidson and Ackerman 1993). A recent estimate indicates that 133 billion tonnes of SOC, which is about 8% of the total global SOC stock, had been lost from the top 2 metres of soil on a global scale since agriculture started about 12,000 years ago, with the rate of loss dramatically increased since the industrial revolution (Sanderman et al. 2017). The Sanderman et al. (2017) study also indicated that the percentage of SOC loss was greater on cropland but the total amount of SOC loss was slightly higher on grazing land as more than twice as much land is grazed. This indicates that there is a greater potential to improve the % SOC gain in cropland but there is a greater potential to increase total SOC storage in grazing land. One of the key aspects of SOC is that the soil and vegetation stores about three times the organic C of the atmosphere (Plate 8.1; FAO 2004), and thus small changes in the organic C stock in the soil and vegetation can cause a large effect on atmospheric CO<sub>2</sub> concentration; therefore, great efforts must be made to increase SOC storage in and to reduce GHG emissions from terrestrial ecosystems. In managed systems, SOC storage can be increased by management practices such as avoiding the burning of crop residues after harvest, and the application of composts and biochar and animal manure to increase organic C input to the soil.

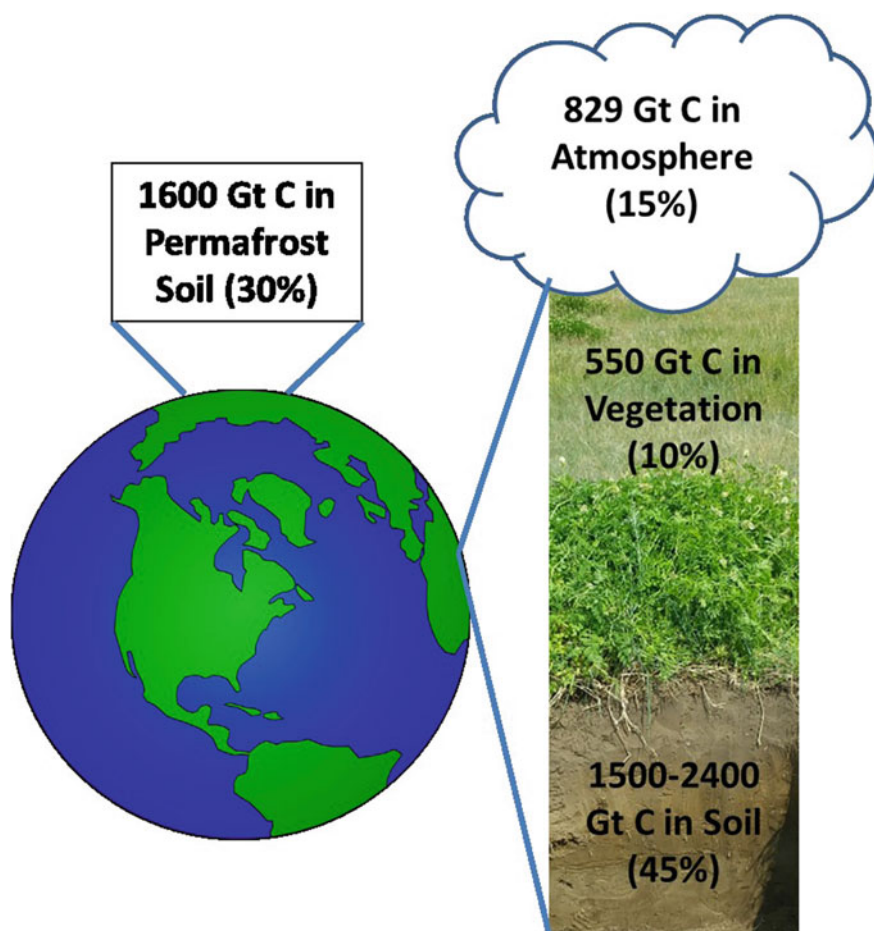
There is significant potential for the agriculture sector to contribute to the reduction of anthropogenic sources of GHG emissions since agriculture is a large source of GHG and to increase soil C storage since much soil C has been lost through cultivation (Plate 8.2; top panel). For example, agricultural management practices can be improved to reduce disturbance to the soil by decreasing the frequency and extent of cultivation as a way to minimise soil C loss and/or to increase soil C storage; if permanent vegetation can be maintained, soil C storage can increase, benefiting from the C cycle becoming more closed in the system and the soil being able to trap more C (Plate 8.2; bottom panel). Fertiliser N use efficiency (NUE) can be improved through strategic application of fertiliser so as to reduce N loss, whether it is through

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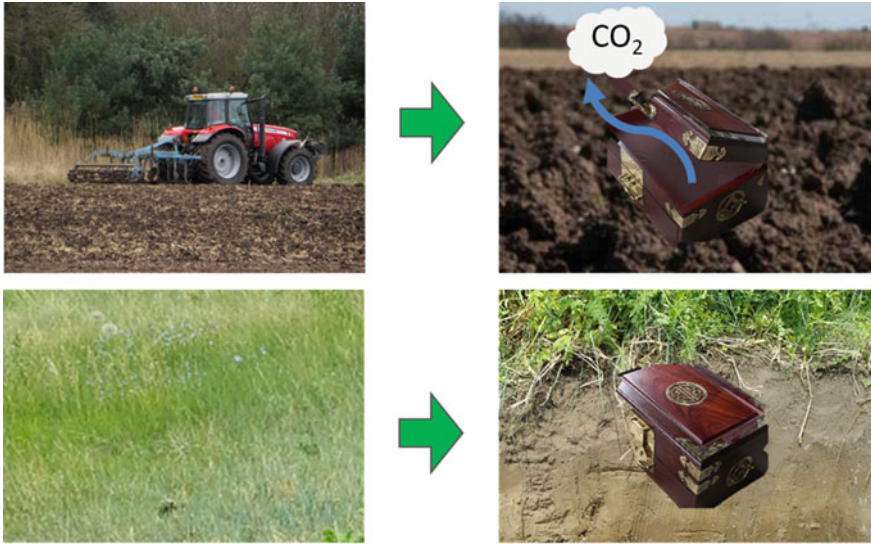
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**Plate 8.1** An illustration of the distribution of carbon in terrestrial ecosystems, including the atmosphere. (Source Schwartz 2014)

leaching or gaseous form of N loss. Management measures can also be taken to minimise agricultural biomass burning. Climate-smart agriculture (CSA) management practices, including strategic use of synthetic and organic fertilisers and water, conservation tillage, use of cover crops, and the use of amendments such as biochar, nitrification inhibitors and lime to agricultural fields, have been considered a way to reduce GHG emissions from agriculture (Bai et al. 2019; Lipper et al. 2014; Zaman et al. 2008a and 2009; Zaman and Blennerhassett 2010). The FAO defines CSA as a systematic approach for developing agricultural strategies that can ensure sustainable food security under predicted climate change scenarios (FAO 2013). Based on this definition, a range of agricultural practices can be developed to help improve food security and environmental quality simultaneously in the context of global change.



**Plate 8.2** An illustration of a disturbance to the soil causes an increased release of  $\text{CO}_2$  and other greenhouse gases (and leading to the soil being leaky, open treasure chest) and in soils with undisturbed vegetation will cause the carbon cycle to be more closed (*Source* FAO 2013)

Since the soil can act as a sink or source for  $\text{CO}_2$  and affect climate change, if we can enhance the C sink strength and remove more  $\text{CO}_2$  from the atmosphere by adopting CSA, then we can be in a win–win situation by not only combating the negative effects of climate change but also improving soil quality and health, including the retention of nutrients and water, and increased agricultural productivity.

## 8.2 Climate-Smart Agricultural Technology to Reduce GHG Emissions

Climate-smart agriculture emphasises on improving risk management, enhancing information flows and promoting local institutions to increase the adaptive capacity of communities to climate change (Campbell et al. 2014), as such CSA plays a pivotal role in maintainable development. Climate-smart agricultural practices such as the use of cover crops, amendments and tillage management play key roles in reducing agricultural GHG emissions. Take  $\text{N}_2\text{O}$ , a potent GHG, as an example, the emission of  $\text{N}_2\text{O}$  is affected by many factors, such as the use of N process inhibitors (urease and nitrification inhibitors), the strategic application of synthetic fertilisers (the right type, the right amount that is based on crop requirement and soil tests, at the right plant growth stage, and the right method (e.g. even spread)), avoiding the application of N fertilisers to very wet or very dry soils, maintaining soil pH above

6 by adding lime to lower the  $N_2O:N_2$  ratio, co-application of animal manure and chemical fertilisers to provide a more balanced nutrient supply, optimising animal stocking rate to avoid over-grazing, keeping animals off the pasture especially in the wet season to minimise N input and avoid soil compaction, and minimising the excessive use of farm machinery. Many of those fall under CSA practices. Below, key CSA practices and their effect on GHG emissions from agriculture are discussed in detail.

### **8.2.1 Nitrogen Process Inhibitors and Greenhouse Gas Emissions**

Multiple microbial soil processes contribute to GHG emissions. One of the major processes contributing to GHG emissions is the mineralization of organic matter by microbial organisms, a process also called soil heterotrophic respiration, where organic C is converted to  $CO_2$  and released to the atmosphere. Nitrification and denitrification are key processes contributing to  $N_2O$  emissions from the soil, where ammonia-oxidation and successive nitrifier denitrification or denitrification, among other processes, can be important pathways for  $N_2O$  emissions (Fig. 7.8) (Guo et al. 2018). In the ammonia oxidation process,  $N_2O$  can be produced by the chemical decomposition of hydroxylamine ( $NH_2OH$ ). One potential way to mitigate  $N_2O$  emissions is to use nitrification and urease inhibitors to slow down the rate of nitrification and reduce the availability of the substrate ( $NH_4^+$ ) for nitrification.

#### **8.2.1.1 Synthetic Nitrification Inhibitors**

Nitrification is the process where ammonium or ammonia is converted to nitrate, via nitrite as an intermediate N species. Since the formation of nitrite is usually the rate-limiting step in the nitrification process, nitrite usually does not accumulate in the soil, unless the soil has a high pH where nitrite oxidisers are inhibited (Rodgers 1986). During the nitrification process,  $N_2O$  can be formed and emitted from the soil. The final product of nitrification, i.e nitrate, is subject to leaching losses, and when anaerobic conditions develop, nitrate is denitrified thus leading to the formation of  $N_2O$ . Both nitrate leaching and denitrification are major pathways for N losses from the soil. Reducing the rate of nitrification can both conserve N in the soil and reduce  $N_2O$  emissions (Abalos et al. 2012). In this respect, the application of nitrification inhibitors can effectively reduce nitrification rates and the buildup of nitrate which can further be denitrified. Nitrification inhibitors are a group of chemical compounds that slow down the conversion rate of ammonium or ammonia to nitrate by inhibiting the ammonia monooxygenase activity, through disrupting the activity of *Nitrosomonas* and *Nitrobacter* bacteria (Abalos et al. 2014). Nitrification



inhibitors are applied with ammonium-, ammonia- or urea-based fertilisers as the application of such fertilisers substantially increases the rate of nitrification. The need for nitrification inhibitors in non-fertilised soils is rare as nitrification rates in such soils are low. The ammonium or ammonia can come from urine, manures, composts or crop residues as they decompose, or fertilisers such as ammonium sulphate or urea (Rodgers 1986). Under suitable conditions (e.g. warm soil temperature and a moisture content near field capacity, when there is still ample O<sub>2</sub> available), nitrification occurs within days or weeks of the application of ammonium-based fertilisers and thus if nitrification can be effectively reduced within that timeframe, N loss from the system can be substantially reduced (Sanz-Cobena et al. 2017).

There are at least eight compounds commercially available as nitrification inhibitors but the most commonly used ones are 2-chloro-6-(trichloromethyl)-pyridine (nitrapyrin), dicyandiamide (DCD), 3,4-dimethylpyrazole phosphate (DMPP, e.g. ENTEC®) (IPNI undated) and pronitradine (Centuro™). Nitrapyrin (commercial product: N-Serve™ and Instinct™) can be applied to the soil in different ways: if anhydrous ammonia is the fertiliser being applied, the nitrification inhibitor can be injected with the fertiliser, if solid N fertilisers are being applied, the inhibitor can be coated onto the fertiliser, and if manure is used as an organic fertiliser, the inhibitor can be mixed with the manure before manure application. Nitrapyrin is usually effective for less than 30 days after being applied to the soil and is volatile, therefore, the best way to apply this inhibitor is to incorporate it into the soil; DMPP can be effective for reducing nitrification rates for 25–70 days and is usually pre-blended with fertilisers; DCD can last 25–55 days and can be coated on solid fertilisers (Sanz-Cobena et al. 2012), or surface applied to soils that have been applied with manure or on grazing land to reduce nitrification from urine patches (IPNI undated). While DMPP is somewhat immobile, DCD can be relatively easily leached from the soil; those behaviours of the nitrification inhibitors should be considered when they are applied in the field.

DCD and DMPP have been found to be equally effective in changing soil inorganic N content, leaching of dissolved inorganic N (DIN) and emissions of N<sub>2</sub>O in a recent meta-analysis of field trials reported in the literature (Yang et al. 2016). Their cost-benefit analysis (CBA) showed that the economic benefit was about seven times greater with DCD than with DMPP when applied with ammonium-based fertilisers to reduce nitrification (Yang et al. 2016). Those two nitrification inhibitors are among the most commonly used. DCD is cheaper and less volatile, but the application rate of DMPP is typically one-tenth that of DCD, and DMPP has a lower eco-toxicological effect for plants as is summarised in Yang et al. (2016). In an Australian study, DMPP application (as ENTEC®) decreased N<sub>2</sub>O emissions by 15% in a subtropical pasture in Queensland (Lam et al. 2018).

The effectiveness of the nitrification inhibitors can be affected by soil properties such as soil water content (Barrena et al. 2017), and soil organic matter and clay content (Zhu et al. 2019). Zhu et al. (2019) reported that the efficiency of DMPP in reducing nitrification and N<sub>2</sub>O emissions was lower in soils with high organic matter



and clay contents, likely due to the high rates of adsorption of DMPP by soil organic matter and clay. The effectiveness of DCD in reducing  $N_2O$  emissions from urine patches in New Zealand is highly season-specific, with reductions of 52, 39 and 16% in autumn, spring and summer, respectively, but DCD application increased  $NH_3$  emissions by 56, 9 and 17% in the respective seasons (Zaman et al. 2009). Management practices can also affect the efficiency of nitrification inhibitors. For example, biochar application to the soil has been shown to decrease the efficiency of DMPP both at 40 and 80% of water-filled pore space (WFPS) in a laboratory incubation study (Fuertes-Mendizábal et al. 2019). The use of nitrification inhibitors increases crop yield and NUE, but the effectiveness was greatest when they are used in coarse-textured soils, irrigated systems and/or crops receiving high rates of N fertiliser input (Abalos et al. 2014).

### 8.2.1.2 Synthetic Urease Inhibitors

Urease inhibitors retard the activity of urease, which exists in the soil and plant residues. Urease is involved in the conversion of urea to ammonium in a process called hydrolysis. Unfortunately, the urea hydrolysis process increases the pH of the soil and causes a large proportion of the formed ammonium to be volatilized as ammonia. Urease inhibitors would slow down the rate of hydrolysis or the rate of release of ammonium, reduce the loss of N as ammonia through volatilization, and increase the NUE of urea fertilisers applied to the soil. One of the main reasons for the improved NUE is for more urea to be washed into the soil over time (when the rate of hydrolysis is suppressed), as urea is highly soluble in water. The better contact with the soil increases the chances of the released ammonium to be adsorbed by the cation exchange sites.

The N-(n-butyl) thiophosphoric triamide (NBPT) and N-(n-propyl) thiophosphoric triamide (NPPT) are two chemicals that have been shown to be effective in inhibiting urease activities (Rodríguez et al. 2019; Lam et al. 2018; Zaman et al. 2009; Sanz-Cobena et al. 2008). Products containing those urease inhibitors include Agrotain™ (that contains NBPT) and Limus™ (that contains both NBPT and NPPT). The application of NBPT prior to urine deposition was more effective in reducing ammonia volatilization loss (17.5–27.6% reduction) as compared with applying the NBPT after the urine deposition (0.6–2.9% reduction) in pastureland in New Zealand (Rodríguez et al. 2019). The effectiveness of NBPT is highly season-specific for reducing  $NH_3$  volatilization loss from urine patches in New Zealand, with reductions of 29, 93 and 31% in autumn, spring and summer, respectively (Zaman et al. 2009). An Australian study reached similar conclusions that urea applied with NBPT (as Green UreaNV®) was effective in decreasing  $NH_3$  volatilization (by 44%) in a subtropical pasture in Queensland (Lam et al. 2018). The effectiveness of NBPT was greater in alkaline soils ( $pH \geq 8$ ) (Abalos et al. 2014). It has also been observed, under laboratory conditions, that application of urease inhibitors in soils where nitrification is the main pathway in the production of  $N_2O$  (i.e. WFPS < 50%), could be an effective way to mitigate these N losses (Sanz-Cobena et al. 2014).

The combined application of urease inhibitor + the nitrification inhibitor, DCD, (e.g. Agrotain Plus) to inhibit both the hydrolysis of urea and the nitrification processes to minimise the N loss showed best results in maximising NUE. When urea and ammonium nitrate (UAN) was applied at 150 mg N kg<sup>-1</sup> in a sandy loam soil in the United States with Agrotain Plus, N<sub>2</sub>O emissions were reduced by 78% compared to the control (Cai et al. 2018). The use of double inhibitors containing both NBPT and DCD (3:7) in a New Zealand study reduced NH<sub>3</sub> volatilization by 14, 78 and 9% in autumn, spring and summer, respectively, and N<sub>2</sub>O emissions by 37, 67 and 28%, respectively, from urine patches (Zaman et al. 2009). However, another study showed that adding DCD with NBPT did not further reduce NH<sub>3</sub> volatilization loss, but in fact enhanced the volatilization loss by maintaining a higher soil NH<sub>4</sub><sup>+</sup> concentration and pH for a longer period of time after urea application, indicating that DCD co-applied with NBPT and urea could offset the effect of NBPT in reducing volatilization losses (Soares et al. 2012).

### 8.2.1.3 Biological Nitrification Inhibitors

Some plant species can release secondary metabolites through root exudation and/or from leaf litter. Such metabolites have the ability to suppress microbial nitrification (Souri and Neumann 2018). In some earlier studies, detectable biological nitrification inhibition (BNI) was found in root exudates of sorghum (*Sorghum bicolor* (L.)), pearl millet (*Pennisetum glaucum* (L.) R. Br.) and groundnut (*Arachis hypogaea* (L.)) among tested cereal and legume crops, while *Brachiaria humidicola* (Rendle) Schweick and *B. decumbens* Stapf had the highest BNI capacity among pasture grass species tested (Subbarao et al. 2007). In addition, when BNI compounds from root exudates were applied to the soil, their inhibitory effects on NO<sub>3</sub><sup>-</sup> formation lasted for more than 50 days (Subbarao et al. 2007). Linoleic acid,  $\alpha$ -linolenic acid, and methyl linoleate, a fatty acid methyl ester of linoleic acid, are some of the example compounds that are effective in biologically inhibiting nitrification (Subbarao et al. 2008). Subbarao et al. (2007) suggested that some level of BNI is likely a widespread phenomenon in tropical pasture grass species and those properties could be used to suppress nitrification in natural or managed systems. The production in and release from plants of BNIs are triggered by the presence of NH<sub>4</sub><sup>+</sup> in the rhizosphere of plants, which means that BNIs are released where the majority of the nitrifier populations reside (Subbarao et al. 2013a). The pH in the rhizosphere will affect the release of BNIs from roots; for example, sorghum plants release BNIs from their roots in the presence of NH<sub>4</sub><sup>+</sup> when the rhizosphere pH is between 5.0 and 6.0 (Subbarao et al. 2013b), indicating the usefulness of BNI to reduce nitrification in alkaline soils will be non-existent or very low. More research is needed to understand and take advantage of BNI in agricultural production systems to reduce N loss and improve NUE.

## 8.2.2 Soil Amendments and Greenhouse Gas Emissions

### 8.2.2.1 Mulch

Addition of mulch to the soil will change the availability of carbon (C) and other nutrients to microbial populations and will thus affect soil GHG emissions. Existence of a litter layer (mulch) can induce microbial N immobilisation in the litter layer, and result in reduced available N and reduced plant growth rate; however, the litter layer may benefit plant growth by conserving soil moisture (Matsushima and Chang 2006) but may reduce soil temperature and N mineralization rates (Matsushima and Chang 2007). Addition of mulch can immobilise mineral N in the soil and reduce the availability of  $\text{NH}_4^+$  for nitrification and  $\text{NO}_3^-$  for denitrification, and thus reduce  $\text{N}_2\text{O}$  emissions as compared to no mulch addition (Wu et al. 2013). Using wood bark mulch reduced the nitrate concentration in the soil and cut soil  $\text{N}_2\text{O}$  emissions by up to 28% in a grape (*Vitis vinifera* L. cv. Merlot) yard on a sandy loam soil in British Columbia, Canada, when measured over a two-year period (Fentabil et al. 2016). However, overloading of straw to the soil surface can delay seed germination and result in the need for additional fertiliser supply to compensate for the N that may be immobilised in the critical period of the early growing season (Procházková et al. 2003). As far as  $\text{CO}_2$  emissions are concerned, mulching will usually result in increased  $\text{CO}_2$  emissions due to the addition of labile C in the mulch, with the rate of  $\text{CO}_2$  emissions increasing with the increased rate of mulch addition (Wu et al. 2013). Major anthropogenic sources of methane emissions include fossil fuel production, landfills and livestock farming, but some agricultural soils can be an anthropogenic source of methane emissions as well (IPCC 2007). In rice paddy systems, straw application has been shown to increase  $\text{CH}_4$  emissions (Bossio et al. 1999; Ma et al. 2008); however, straw addition significantly reduced  $\text{CH}_4$  emissions under an aerobic condition in a laboratory incubation experiment, indicating that under upland conditions, straw application increased the soil's ability to take up  $\text{CH}_4$  (Tate et al. 2007).

### 8.2.2.2 Biochar

Biochar has been widely studied for its effects on GHG emissions. Biochar addition to the soil can change a range of soil properties, including the cycling of C and N. Biochar application has been widely reported to reduce  $\text{N}_2\text{O}$  emissions (Wu et al. 2013; Cayuela et al. 2013; Chang et al. 2016; Hüppi et al. 2016). The application of biochar to the soil has been shown to reduce denitrification and decrease  $\text{N}_2\text{O}$  emissions by 10–90% when tested on 14 different agricultural soils, where a consistent reduction of the  $\text{N}_2\text{O}/(\text{N}_2 + \text{N}_2\text{O})$  ratio was observed, indicating that biochar reduces  $\text{N}_2\text{O}$  emissions by facilitating the last step of the denitrification process and producing more  $\text{N}_2$  rather than  $\text{N}_2\text{O}$  (Cayuela et al. 2013). However, in some soils,

biochar application can stimulate nitrification and increase  $N_2O$  emissions; therefore, the effect of biochar application on  $N_2O$  emissions is related to the dominant  $N_2O$  formation pathway that operates in a soil (Sánchez-García et al. 2014).

Biochar application has been reported to reduce  $CH_4$  emissions from paddy soils, one of the largest anthropogenic sources of  $CH_4$  emissions on a global scale (Feng et al. 2012). The reduction in paddy  $CH_4$  emissions by biochar application was not a result of the inhibition of methanogenic archaea, but resulted from increased methanotrophic proteobacterial abundances and decreased ratios of methanogenic to methanotrophic abundances (Feng et al. 2012).

In agricultural production systems, a large quantity of crop residues are produced and the return of crop residues in the raw form vs after the crop residue is converted to biochar can have dramatic effects on the emissions of all three trace gases (Wu et al. 2013), and most research suggests that there are substantial beneficial effects to be gained in mitigating climate change by converting crop residues to biochar and applying biochar to the soil instead. The effect of biochar on GHG emissions itself is highly complex as many factors alter the function of biochar on C and N transformation processes and thus GHG emissions. When the effect of the stable C input in the form of biochar is considered, the application of biochar is considered an effective technique to mitigate climate change due to its negative emissions potential ( $0.7 \text{ Gt Ceq. Yr}^{-1}$ ) and its lower impact on land, water use, albedo, energy requirement and cost as compared to other negative emissions technologies such as direct air capture, increased weathering that takes up  $CO_2$  from the air, bioenergy projects with C capture and storage, and afforestation/deforestation (Smith 2016).

### 8.2.2.3 Liming to Shift the Balance Between $N_2O$ and $N_2$ Emissions

Soil pH is one of the key regulators of microbiological processes that affect  $N_2O$  and  $N_2$  production and their ratio. The soil pH threshold for nitrification is 5; however, nitrification can occur even below pH 5 as some nitrifier strains are adapted to acidic conditions (Bouwman 1990). Denitrification has been reported to occur over a wide range of soil pH (5–8) (Flessa et al. 1998); however, laboratory experiments with artificially adjusted soil pH suggest that, under optimised conditions (very low  $pO_2$ ,  $NO_3^-$  and with glucose amendment), denitrification can proceed even at pHs below 4 or above 10 (Šimek et al. 2002). Numerous laboratory and field studies have shown that soil pH affects  $N_2O$  and  $N_2$  emissions and thus the ratio of these gases (e.g. Stevens and Laughlin 1998). In experiments conducted under controlled environmental conditions, raising soil pH to 7 through lime application has been found to significantly increase  $N_2$  emissions from pasture and wetland soils treated with cow urine, urea and  $KNO_3$  at  $200 \text{ kg N ha}^{-1}$  (Zaman et al. 2007, 2008b). Similar trends of enhanced  $N_2$  emissions after raising soil pH to 7 was observed in pasture soils treated with urea/urine in a field experiment (Zaman and Nguyen 2010). In another study, a site with the greatest animal impact, the ratio of  $N_2$  to  $N_2O$  produced during denitrifying enzyme activity (DEA) measurements was fivefold higher, and the pH

was 2 units higher, than a site with the least animal impact, indicating that soil conditions were favourable for production of  $N_2$  rather than  $N_2O$  in the area with intense excretal returns and treading (Hynšt et al. 2007).

Most researchers attribute high  $N_2O$  and low  $N_2$  emissions in acidic conditions to the suppression of  $N_2O$ -reductase (inhibition starts at soil pH 4.5) (Daum and Schenk 1998; Flessa et al. 1998; Stevens and Laughlin 1998; Zaman et al. 2007). It is also possible that denitrifying enzymes are susceptible at low soil pH and produce  $N_2O$  from intermediate products (Nägele and Conrad 1990). However, the lower rates of  $N_2$  emissions and higher  $N_2O:N_2$  ratio at low soil pH could be due to lower amounts of soil organic C and mineral N available to the denitrifying population rather than a direct effect of low pH on denitrification enzymes (Šimek and Cooper 2002). Regardless of the biochemical mechanism for soil pH effects on  $N_2$  emissions, raising soil pH through the application of amendments such as lime appears a viable approach to mitigate  $N_2O$  emissions (Šimek et al. 2002; Zaman and Nguyen 2010; Zaman et al. 2007, 2008b).

### ***8.2.3 Fertiliser Type and Management and Greenhouse Gas Emissions***

The use of different fertiliser types and the management of fertiliser applications can have marked effects on nitrification, denitrification and GHG emission rates (Mosier et al. 2006; Wang et al. 2018). As discussed in the earlier section, the type of fertiliser (some fertilisers are acid-forming while others raise soil pH when applied to the soil) applied can affect the total amount of GHG emitted as well as the  $N_2O:N$  ratio in the emissions. A number of agronomic practices have been widely tested to minimise N losses from agricultural production systems, for example, alteration of the rate or timing of fertiliser application, such as autumn vs. spring, basal vs. broadcast, deep vs. surface applications, point injection placement of solutions, foliar applications of urea (Subbarao et al. 2013b), and split application. Fertilisers such as polythene-coated urea (PCU) that releases N slowly in the soil have been demonstrated to reduce nitrification (Zvomuya et al. 2003); however, the use of such fertilisers can be limited by the high cost for purchasing such fertilisers (Subbarao et al. 2013b).

### ***8.2.4 Cropping Systems and Greenhouse Gas Emissions***

#### ***8.2.4.1 Agroecosystems***

The type of cropping system used has a significant effect on GHG emissions as cropping systems will differ in their fertilisation regime, crop productivity (and thus the amount of organic matter input to and retention in the soil), crop species being

used (N-fixing vs. non-N-fixing crop species), water management practices, and tillage management, among others (Snyder et al. 2009). On an extreme case, GHG emission rates and timing will be dramatically different between rice cultivation in flooded fields vs. wheat production in well-drained upland sites in a rice-wheat rotation, where the rice-wheat belt makes up 24–27 million ha in South and East Asia (Wassmann et al. 2004).

Rice cultivation in paddy fields is a unique cropping system in Southeast Asia. In Vietnam, for example, rice production is the largest source of agricultural GHG, with 37.4 Tg CO<sub>2</sub> equivalent of total emissions, that account for 58% of agricultural GHG emissions in that country (United Nations 2013). Many factors, such as the management of fertiliser applications, animal manure and crop residue management, water regime used during rice production, and use of urease and/or nitrification inhibitors, will affect the emissions of GHGs from paddy fields. One of the biggest concerns of GHG emissions from paddy fields is the emission of CH<sub>4</sub> as paddy fields are mostly submerged in water during the growing season and anaerobic condition caused substantial CH<sub>4</sub> emission to occur. The CH<sub>4</sub> emissions are the balance of CH<sub>4</sub> production and oxidation in the soil, and are affected by factors influencing the transportation of CH<sub>4</sub> from the anoxic soil/free-standing water to the atmosphere (Aulakh et al. 2001). Up to 80% of the CH<sub>4</sub> produced in paddy soils is oxidised in the rhizosphere or microsites that are less anaerobic before it is released to the atmosphere (Sass et al. 1991; Holzapfel-Pschorn et al. 1985) and this process helps to dramatically reduce the rate of CH<sub>4</sub> emissions from paddy soils.

Proper water management can substantially reduce CH<sub>4</sub> emissions from paddy fields; midseason drainage has been reported to reduce CH<sub>4</sub> emissions by 44% and alternating wetting–drying cycles at ten-day intervals by 61% as compared to continuously flooded management in southeast China (Lu et al. 2000). Application of urea can increase CH<sub>4</sub> emissions from paddy fields as ammonium can inhibit CH<sub>4</sub> oxidation (Conrad and Rothfuss 1991). However, others found that application of ammonium-based fertilisers can reduce CH<sub>4</sub> emissions as ammonium enhances methanotrophic bacteria activities in the rhizosphere of rice plants (Bodelier et al. 2000). The effect of ammonium on CH<sub>4</sub> oxidation is dependent on the CH<sub>4</sub> concentration: inhibition at low initial CH<sub>4</sub> concentration (500 μl l<sup>-1</sup>) but stimulation at high initial CH<sub>4</sub> concentration (2000 μl l<sup>-1</sup>) (Cai and Mosier 2000). However, ammonium sulphate has been found to be a promising fertiliser to use, as opposed to urea, to mitigate CH<sub>4</sub> emissions as sulphate enhances sulphate-reducing bacteria activities, which decrease the availability of substrates for methanogens in submerged soils (Yagi et al. 1997). Reductions in CH<sub>4</sub> emissions after ammonium sulphate application has been reported to range from 10 to 67% (Schütz et al. 1989; Wassmann et al. 2000). Ammonium is the preferred N form for rice, therefore, applying ammonium form of N that does not enhance CH<sub>4</sub> emissions would be beneficial for the environment. Application of sulphate-containing phosphorus fertilisers (Achnich et al. 1995) and gypsum (CaSO<sub>4</sub>) (Lindau und Bollich 1993), a common soil amendment for sodic and/or alkaline soil reclamation, have been shown to reduce CH<sub>4</sub> emissions

from paddy soils. Therefore, choice of fertiliser is important in minimising CH<sub>4</sub> emissions from rice paddies, in this case, the use of sulphate-based N or phosphorus fertilisers would be preferred in flooded rice fields.

Applying organic fertilisers such as animal manure and returning crop residue to the soil are necessary to maintain and enhance the sustainability of rice production. However, addition of organic materials to rice fields would increase CH<sub>4</sub> emissions, and organic materials application to the soil can lower the soil redox potential and supply C to methanogens that are responsible for CH<sub>4</sub> production in paddy soils. The production of CH<sub>4</sub> in paddy soils is markedly influenced by the quality and quantity of organic materials added to the soil (Minasny et al. 2017). Therefore, proper manure and crop residue management strategies need to be developed to achieve environment-friendly rice cultivation. One potential alternative to the application of crop residues to paddy soil is to convert crop residues to biochar that is slow to decompose after soil application (Ippolito et al. 2012); biochar application to soil can increase soil aeration and soil C content but mitigate CH<sub>4</sub> emissions as compared with the conventional crop residue application (Feng et al. 2012; Karhu et al. 2011; Liu et al. 2011; Xie et al. 2013). However, other studies reported increased CH<sub>4</sub> emissions from rice fields after biochar application, which may be related to increased substrate supply and enhanced environment for methanogenic activity, and increased rice growth (Knoblauch et al. 2011; Lehmann et al. 2011). The effects of biochar application on CH<sub>4</sub> emissions is thus dependent on soil type, agricultural management practices used, and the type of biochar applied (Lehmann et al. 2011; Waters et al. 2011). Site-specific research should be conducted before any recommendation on agricultural management practices is made for farmers to adopt.

In rice paddies, the flooded condition is conducive for denitrification to occur. Even though less attention has been paid to N<sub>2</sub>O as compared to CH<sub>4</sub> emissions from paddy fields, N loss in the form of NH<sub>3</sub> volatilisation and N<sub>2</sub>O emissions and NO<sub>3</sub><sup>-</sup> leaching affect GHG emissions as well as NUE in rice production systems. Therefore, urease inhibitors can be used to slow down the rate of urea hydrolysis when urea is used as the main N fertiliser (Rogers et al. 2015), while the use of nitrification inhibitors can reduce nitrate leaching loss (Li et al. 2008) and N<sub>2</sub>O emissions (Majumdar et al. 2000; Kumar et al. 2000).

### 8.2.4.2 Organic Farming

Organic farming has long been considered a viable agricultural practice to improve soil health, reduce the resource use intensity, reduce the environmental impact of agriculture and improve food quality (Squalli and Adamkiewicz 2018). The comprehensive study of Squalli and Adamkiewicz (2018), based on longitudinal state-level data in the United States collected between 1997 and 2010, demonstrates that a 1% increase in organic farming acreage can result in a 0.049% reduction in GHG emissions; however, they showed that the net effect of organic farming on GHG emissions is dependent on the contribution of transportation (fuel burning) on methane and nitrous oxide emissions, even though their calculation indicates that the negative



environmental effect of transportation output associated with organic food production is small relative to the environmental benefits of organic farming. A study in Switzerland showed a 40% reduction in  $N_2O$  emissions for organic compared to conventional systems; however, yield-scaled  $N_2O$  emissions under silage maize were not different between organic and conventional systems (Skinner et al. 2019), indicating that even if we consider the lower yield in organic systems, organic farming does not have a negative effect on GHG emissions. In contrast, Aguilera et al. (2015) and Cayuela et al. (2017) showed in two Meta-analyses that fertilisation with solid organic manures (the most used form of N fertilisers in organic agroecosystems) led to the lowest  $N_2O$  emission factor in Mediterranean cropping systems worldwide. On the other hand, we must recognise that organic farming does have a lower crop yield (c. 25% on average) which may require a larger area of land to be cultivated to produce the same amount of food compared to conventional farming (Kniss et al. 2016).

#### 8.2.4.3 Row, Intercropping and Crop Rotation

When dealing with a cropping system that involves rice production (e.g. in Southeast Asia), it is often difficult to strike a balance in the mitigation of different GHGs. Using a rice-wheat crop rotation system as an example, the two crops markedly differ in their nature and intensity of GHG fluxes, where  $CH_4$  emissions are a major contributor to GHG emissions from rice paddies. In rice production systems, water regimes, rice cultivars and soil properties all markedly affect  $CH_4$  emissions (Cai et al. 2003). In addition,  $N_2O$  is also emitted in large quantities from rice production systems following aerobic-anaerobic cycles; on the other hand,  $N_2O$  is emitted in short-term pulses after fertilisation, heavy rainfall or irrigation events and is the main GHG emissions of concern in upland wheat production systems (Wassmann et al. 2004). It is often difficult to balance emissions between  $CH_4$  and  $N_2O$  when designing GHG mitigation strategies in a rice-wheat system, as measures to reduce  $CH_4$  emissions often intensify  $N_2O$  emissions (Wassmann et al. 2004).

In evaluating the effect of intercropping on GHG emissions, Ricord (2018) studied GHG emissions from a sole maize crop, a sole soybean crop and a maize-soybean intercrop and found that the cereal-legume intercropping system effectively reduced  $N_2O$  emissions. In a similar study on the North China Plain,  $N_2O$  fluxes were lower from maize-soybean intercropping than a maize monoculture system in three growing seasons (2013–2015), when all cropping systems were applied with  $240 \text{ kg N ha}^{-1}$  as urea in two split applications (Shen et al. 2018). Shen et al. (2018) showed that the fertiliser N loss as  $N_2O$  was lower in the maize-soybean intercropping (1.6%) and soybean monoculture (1.7%) than in the maize monoculture (2.3%), concluding that maize-soybean intercropping should be recommended as a climate-smart cropping systems for use on the North China Plain. A maize-wheat intercropping system coupled with reduced tillage and stubble mulching can increase grain production and

decrease C emissions in an arid area in northwest China (Hu et al. 2015; Yin et al. 2018). Therefore, choice of a cropping system to use and the associated management practices are important decisions to make to minimise GHG emissions.

### 8.3 Climate-Smart Agriculture (CSA) Practices and C Sequestration

Climate-smart agriculture has three key objectives: (1) to increase agricultural production per unit land area so as to increase income, food security and community development, (2) to improve the adaptive capacity at multiple levels, i.e. from the farm to the national level and (3) to reduce GHG emissions and to enhance C sinks in ecosystems (Campbell et al. 2014). Climate-smart agricultural management practices, including conservation tillage, use of cover crops, and biochar application to agricultural fields, has been widely considered a way to reduce GHG emissions from agriculture.

Conventional tillage has been identified as one of the causes of widespread land degradation problems, such as deterioration of soil structure, soil erosion and decreased soil fertility, affecting the long-term sustainability of agricultural production (Barber et al. 1996). Many climate-smart agricultural technologies have been tested to improve SOC storage in the agricultural landscape, and many of those have been demonstrated to be effective. Climate-smart agricultural technologies such as the use of cover crops, use of perennial crops, application of manure and biochar, reduced/minimum tillage or zero tillage, and crop rotation have all been shown to increase SOC storage.

Field experiments in Australia on light-textured soils in southern Australia indicate that conservation tillage (3–19 years in duration) was effective in increasing SOC levels as compared with conventional tillage, but only in areas with >500 mm annual precipitation and in the top 2.5–10.0 cm of the soil; the lack of conservation tillage effects on SOC levels in other climatic condition or soil layers was mainly attributed to low crop yield related to low rainfall, partial removal of stubble due to grazing and the high decomposition rate in areas with high air temperature (Chan et al. 2004).

### 8.4 Life Cycle Assessment (LCA) for Estimating the C Footprint of Agro-Food Systems

The proposal of effective GHG mitigation strategies in the agri-food sector needs to be based on a whole-system approach. This means that not only direct emissions but also indirect GHG losses (both upstream and downstream from the production

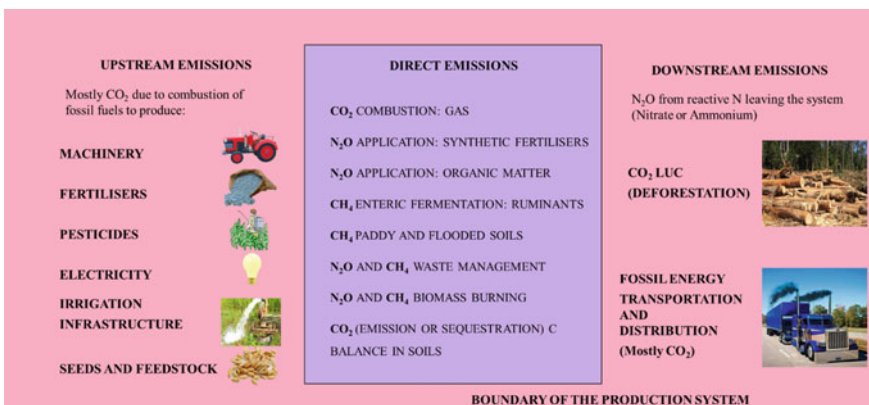
systems) must be considered. For that purpose, the calculation and use of the “C footprint” (CF) or “C budget” of agri-food entities, from products to systems, is primordial (highly important) (Plate 1.1).

The calculation of the CF of the agri-food system requires the accounting for GHG emissions that occur in each of the phases of food and feed production, including not only those that take place in the agricultural sector itself, that is, in crop fields and farms, but also during the manufacture of agricultural inputs, or those derived from the distribution, marketing and consumption of food, using an LCA approach.

The LCA adopts a “bottom-up” approach, that is, it records in detail the emissions generated along the product supply chain, using information on production technologies of the different goods and services. To this end, a “product system” has to be defined, which includes both the different phases of the supply chain (i.e. the “life cycle” of the product) and the exchanges that occur with the environment (i.e. GHG emissions); and a “functional unit” for each food (e.g. 1 kg of product). As a result of the application of an LCA, the “emission coefficients” (i.e. the amount of GHG emissions in kg CO<sub>2</sub> eq./kg of product) are obtained, which can be applied to both intermediate and final products, whether domestic or imported.

In estimating the CF of the Spanish Agri-food sector, Aguilera et al. (2015) used the following information as a source (Plate 8.3): (i) inventory analysis based on previous work of the Laboratory of History of Agroecosystems (UPO), based on official data (Yearbook of Agricultural Statistics, FAOSTAT, National Emissions Inventory, etc.); (ii) industrial input emission factors based on “embodied energy” (Aguilera et al. 2015); (iii) Mediterranean N<sub>2</sub>O emission factors–meta-analysis (Cayuela et al. 2017); (iv) C sequestration with HSOC model (Aguilera et al. 2018); and (v) calculation of emissions associated with irrigation (Aguilera et al. 2019).

National inventories of atmospheric emissions, prepared by the signatory countries to verify compliance with the Kyoto Protocol, is the IPCC Tier 1 approach that is based on global emission factors. However, there is growing evidence that the



**Plate 8.3** An example of processes considered, and the main sources used to estimate the carbon footprint of the Spanish agri-food sector. Reproduced courtesy of Aguilera (2015)

factors differ depending on the type of climate, soil conditions and management, so it is necessary to use more specific factors for a more accurate assessment of emissions. The balance of C in the soil is a crucial process in the CF of agricultural products, since as we have seen it can compensate a large part of the emissions, when the soil gains organic matter and behaves as a sink; otherwise, the soil would contribute more to GHG emissions, when the soil loses organic matter. Despite this, it has been shown that, particularly under arid and semi-arid conditions, C in the soil is very sensitive to changes in management regimes, with changes that can range from the loss of half a ton of C per year in unfertilised soils, to the gain of more than one ton of C in soils in which organic amendments are applied.

The emissions derived from the production of agricultural inputs have decreased significantly due to improvements in efficiency in the industry; therefore, it is necessary to use temporal dynamic factors for the evaluation of historical GHG emissions. Quantitative reviews have been published that analyse precisely these factors for conditions comparable to those in European countries (Aguilera et al. 2015).

The sources of GHG emissions include the construction and maintenance of the agricultural infrastructure, the direct and indirect emissions associated with the use of energy, including traction animals, power generation and fuel use, and CH<sub>4</sub> emissions from water bodies (reservoirs, rafts, ditches and canals) (Aguilera et al. 2019). Emissions related to the use of energy could be estimated considering the changes in the country's electric mix, in the energy efficiency of electric generation, and in fossil fuel extraction techniques, including associated methane emissions. The GHG emissions associated with water bodies should include CH<sub>4</sub>, CO<sub>2</sub> and N<sub>2</sub>O.

Given the challenges that we face in reducing GHG emissions at the agri-food system level, it is necessary to advance our knowledge about effective mitigation strategies that are adapted to the soil-climatic conditions in each region, for example, by synthesising the existing relevant information regarding the main agricultural management practices and their impact on the mitigation of GHG emissions, C sequestration, other polluting compounds, as well as potential barriers and opportunities for the implementation of these strategies.

## 8.5 Conclusions

We conclude that CSA practices, with an emphasis on climate change adaptation and mitigation, can take many different forms. The CSA practices have many roles to play in agricultural sustainability and in reducing GHG emissions, as well as in increasing soil C sequestration. Practices such as the use of nitrification and urease inhibitors, mulching, application of biochar to the soil, fertilisation management and use of intercropping and crop rotations are all options available to landowners to effectively adapt to and mitigate regional to global climate change. The reader is, however, cautioned that the best CSA practice to be applied to a specific system or

location depends on many different factors. Region- or site-specific research is often needed prior to their application to determine if any of the CSAs might produce a positive result on climate change adaptation and mitigation.

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