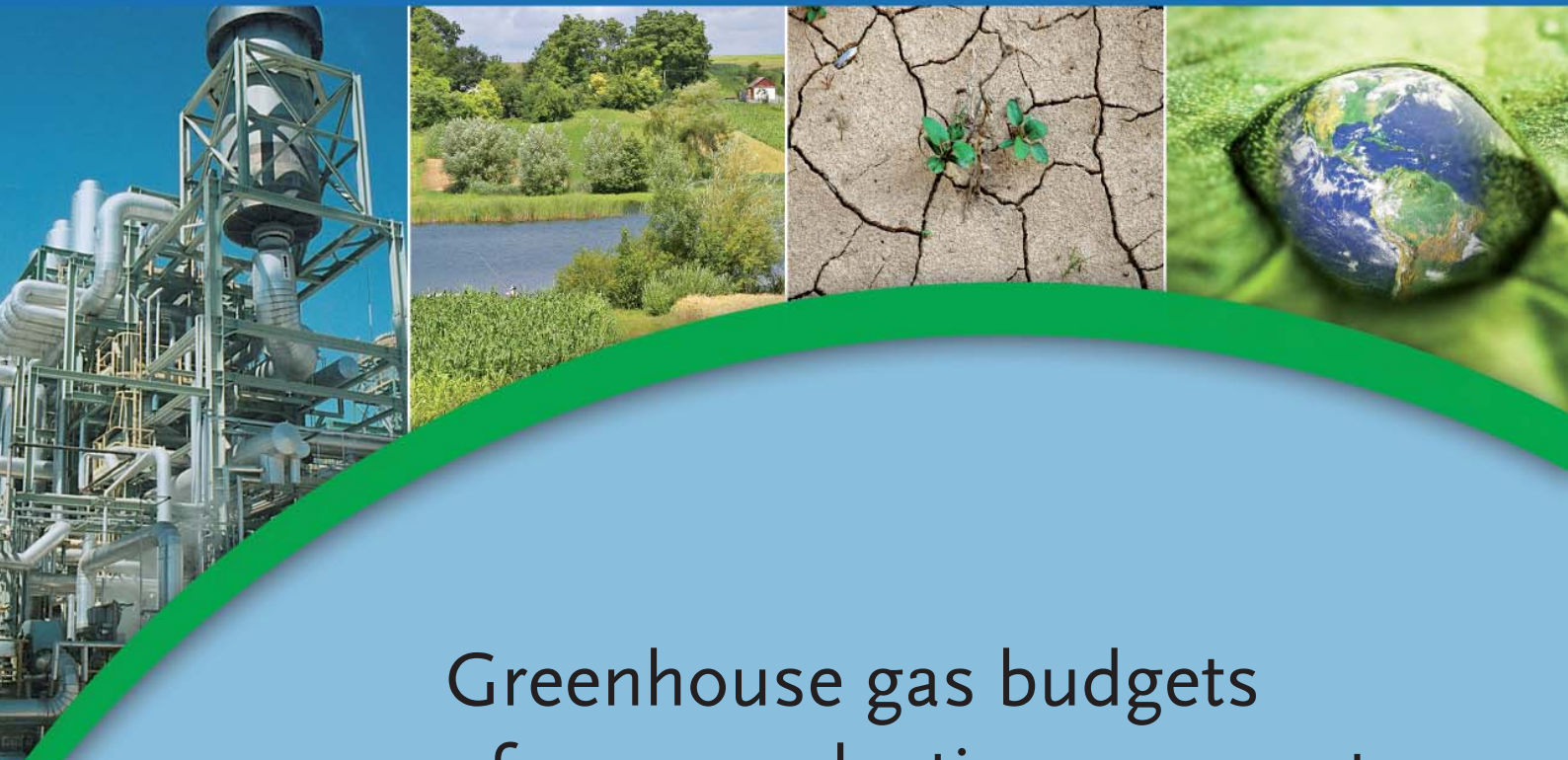




International  
Fertilizer Industry  
Association



# Greenhouse gas budgets of crop production – current and likely future trends

Helen C. Flynn and Pete Smith

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Helen C. Flynn and Pete Smith

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## About the report and the authors

### About the report

This report has been prepared on request of the International Fertilizer Industry Association (IFA), as a contribution to the international negotiations on climate change. It provides an up-to-date state of the scientific knowledge on greenhouse gas balances in relation to crop production and fertilizer use. This report has been written for policy makers, scientists and the fertilizer industry.

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## Symbols, units, acronyms and abbreviations

(as used in this report)

BAT	Best available technology
BMP	Best management practice
C	Carbon
CH <sub>4</sub>	Methane
CI	Confidence interval
CO <sub>2</sub>	Carbon dioxide
CO <sub>2</sub> -eq	Carbon dioxide equivalent
DCD	Dicyandiamide
DNDC	DeNitrification-DeComposition
ECCP	European Climate Change Programme
EF	Emission factor
EU	European Union
FAO	Food and Agriculture Organization of the United Nations
Gcal	Gigacalorie (10 <sup>9</sup> calories)
Gg	Gigagram (10 <sup>9</sup> grams = 1 thousand metric tonnes)
GHG	Greenhouse gas
GWP	Global warming potential
ha	Hectare
IFA	International Fertilizer Industry Association
IPCC	Intergovernmental Panel on Climate Change
K	Potassium
kg	Kilogram
km	Kilometre
LCA	Life cycle assessment
LUC	Land use changes
Mha	Million hectares
mV	Millivolt
N	Nitrogen
N <sub>2</sub> O	Nitrous oxide
NH <sub>3</sub>	Ammonia
NH <sub>4</sub> <sup>+</sup>	Ammonium
NUE	Nitrogen use efficiency
P	Phosphorus
Pg	Petagram (10 <sup>15</sup> grams = 1 billion metric tonnes)
SOC	Soil organic carbon
t	Metric tonne
Tg	Teragram (10 <sup>12</sup> grams = 1 million metric tonnes)
UNFCCC	United Nations Framework Convention on Climate Change
USA	United States of America
US-EPA	United States Environmental Protection Agency
yr	year

## Executive summary

Agriculture contributes around 10-12 % of total global greenhouse gas (GHG) emissions but is the main source of non-carbon dioxide (CO<sub>2</sub>) GHGs, emitting nearly 60 % of nitrous oxide (N<sub>2</sub>O) and nearly 50 % of methane (CH<sub>4</sub>) (Smith *et al.*, 2007a). N<sub>2</sub>O is produced by microbial transformations of nitrogen (N) in soils and animal waste and therefore often associated with N fertilizer inputs in agricultural systems. CH<sub>4</sub> is generated when organic matter decomposes under anaerobic conditions and is mainly associated with ruminant livestock, manure storage and rice production under flooded conditions. These emissions are currently estimated as 3.3 Pg CO<sub>2</sub>-eq yr<sup>-1</sup> from CH<sub>4</sub> and 2.8 Pg CO<sub>2</sub>-eq yr<sup>-1</sup> from N<sub>2</sub>O emissions (Smith *et al.*, 2007a). Large exchanges of CO<sub>2</sub> occur between the atmosphere and agricultural ecosystems but emissions are thought to be roughly balanced by uptake, giving a net flux of only around 0.04 Pg CO<sub>2</sub> yr<sup>-1</sup>, less than 1 % of global anthropogenic CO<sub>2</sub> emissions (Smith *et al.*, 2007a). However, land use change towards more cultivated land may contribute a further 5.9 ± 2.9 Pg CO<sub>2</sub>-eq yr<sup>-1</sup>, representing 6-17 % of total global GHG emissions (Bellarby *et al.*, 2008), and if indirect emissions from agrochemical and fuel usage are also included, an extra 0.4-1.6 Pg CO<sub>2</sub>-eq yr<sup>-1</sup> (0.8-3.2 %) can be attributed to agriculture. In total, direct and indirect emissions from agricultural activity and land use change to agricultural use could contribute around a third of all GHG emissions (Bellarby *et al.*, 2008).

Globally, agricultural land use has increased by 0.8 % between 1991 and 2002, and these changes are split with an increase of 2.1 % in developing countries partially mitigated by a 1.5 % drop in the developed world (Smith *et al.*, 2007a). This trend is likely to continue with projected increases in world population, and shifts in diet requiring more resources per unit of food produced, being concentrated in areas such as South and East Asia. If agricultural production is going to significantly increase while also minimizing its impact on future climate change, it is important to understand both its current contribution to GHG budgets and how agricultural management practices can influence them. This report explores these issues and identifies key gaps in our knowledge and problems which are setting back our understanding. These include the lack of work addressing GHG emissions on the basis of agricultural productivity rather than

cultivated area, and inconsistent methodologies for measuring things like soil carbon under different tillage regimes and for calculating N<sub>2</sub>O emissions, which make comparisons between systems difficult. There is also a distinct lack of research covering tropical regions, a gap which needs to be urgently addressed given the likely increases in production in these regions. This is especially important because the current trend, for example in Latin America, is towards increasing areas of cultivation rather than intensifying production on existing agricultural land (van Vuuren *et al.*, 2008). This will have a disproportionately large impact on GHG budgets due to the loss of stored soil organic carbon (SOC) which occurs when forests and grasslands are converted to cropland (Murty *et al.*, 2002; Guo and Gifford, 2002; Carlton *et al.*, 2009).

If agriculture continues to develop according to existing trends and no action is taken to mitigate GHG emissions from the sector, they are expected to reach around 8.2 Pg CO<sub>2</sub>-eq yr<sup>-1</sup> by 2030 (Smith *et al.*, 2007a; Vergé *et al.*, 2007). However, there is significant potential to mitigate these emissions using existing agricultural technology. Estimates of this potential vary, especially when economic considerations are included in the calculations, but around 1.5-4.3 Pg CO<sub>2</sub>-eq yr<sup>-1</sup> seems reasonable, with the greatest potential lying in cropland management practices (Smith *et al.*, 2007a). Of these practices, improving nutrient management is particularly crucial, especially given the need to increase agricultural productivity while cultivating as little new land as possible. Key to this is improving crop N use efficiency (NUE) through the use of fertilizer best management practices (BMPs); using the right source, at the right rate, at the right time, and at the right place (Roberts, 2007). Implementation of fertilizer BMPs has been shown to both reduce N applications and associated N<sub>2</sub>O emissions and increase yields. For example, in China, the world's largest consumer of mineral N fertilizers, BMPs have been shown to reduce N inputs by 20-40 %, increase yields by 2-12 %, increase N recovery rates by 10-15 % and reduce N losses by 10-50 %, in comparison with traditional farming practices (Zhang *et al.*, 2007). Even in developed countries with existing trends of improving NUE, there is still the potential for further mitigation (ECCP, 2001; Halvorson *et al.*, 2009; US-EPA, 2009).

Better integration of organic resources such as animal waste and crop residues into crop nutrition programs can assist in improving soil fertility while also helping to mitigate indirect emissions from fertilizer production. These indirect emissions currently contribute around 420 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> (IFA, 2009a) and there is considerable scope to mitigate these, and any future increases, using existing methods such as carbon capture and N<sub>2</sub>O abatement technologies. This could save around 200 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> (IFA, 2009a). Other GHG mitigation strategies include the use of no-till or reduced tillage regimes. There is debate regarding the mitigation potential of tillage measures. This is because assessing the net impact on GHG emissions requires comparing the impacts on both SOC, which is often biased by field measurements taken only in the top 30 cm of the soil profile (Baker *et al.*, 2007), and N<sub>2</sub>O emissions, which are highly variable over time. The balance of evidence does, however, point to a net benefit for suitable soil types, although more research may still aid our

understanding in this area. Reducing tillage also gives indirect savings in terms of reducing on-farm fuel use and associated emissions.

Agronomy measures are perhaps the most difficult mitigation practices to assess at present. Using catch crops, legumes and particular types of crop rotations could potentially reduce GHG emissions per hectare of cropland but can also impact on yields, potentially requiring additional land to be cultivated at great cost in terms of SOC losses. For example, the global warming potential (GWP) of an intensive continuous maize crop may be 2-3 times higher, on a per hectare basis, than that of a conventionally-tilled maize-wheat-soybean rotation, but produce only 63 % of the net GHG emissions when compared on the basis of CO<sub>2</sub>-eq per Gcal of food yield (Robertson *et al.*, 2000; Adviento-Borbe *et al.*, 2007; Snyder *et al.*, 2009). Therefore, more work is needed to compare net GHG emissions from different cropping systems over the long term and on a per unit of production basis.



## 1. Introduction

Agriculture contributes around 10-12 % of total global greenhouse gas (GHG) emissions but is the main source of non-carbon dioxide (CO<sub>2</sub>) GHGs, emitting nearly 60 % of nitrous oxide (N<sub>2</sub>O) and nearly 50 % of methane (CH<sub>4</sub>) (Smith *et al.*, 2007a). CO<sub>2</sub> is mostly released when microbes decompose plant or soil organic matter under aerobic conditions, or when organic matter is burnt, but uptake by vegetation means that the net flux accounts for less than 1 % of global anthropogenic CO<sub>2</sub> emissions (Smith *et al.*, 2007a). N<sub>2</sub>O is produced by microbial transformations of nitrogen (N) in soils and animal waste, under both aerobic and anaerobic conditions, and emissions are often highest when available N exceeds plant requirements, especially under wet conditions (Oenema *et al.*, 2005; Smith and Conen, 2004). CH<sub>4</sub> is generated when organic matter decays under anaerobic conditions, the main examples being fermentation digestion by ruminant livestock, microbial decomposition of stored manure, and rice production under flooded conditions (Mosier *et al.*, 1998b). Emissions from land use changes and fuel and energy use are all accounted for separately within the Intergovernmental Panel on Climate Change (IPCC) methodology, but if changes to cultivated land and indirect emissions associated with agriculture, such as fuel use for farm vehicles and for agrochemical and fertilizer production, are also included, then agriculture could contribute around a third of total global GHG emissions, mainly due to deforestation for agriculture (Bellarby *et al.*, 2008).

The world population has doubled over the past 40 years and is predicted to reach 9 billion by 2040 (US Census Bureau, 2008), putting increasing demands on agricultural production. As a result, the US-EPA (2006a) estimates that GHG emissions from agriculture will increase by around 10-15 % per decade over the next 30 years. However, emissions are decreasing in some regions, such as Europe, and active management of agricultural systems has the potential to mitigate significant levels of GHG emissions, helping to limit the impact on future climate.

This review focuses on the GHG budgets of crop production, in terms of N<sub>2</sub>O, CH<sub>4</sub> and CO<sub>2</sub>, and the impact of fertilizer use on them. It does not consider ammonia or other N oxides, and only briefly touches on the contribution of livestock production and

manure recycling to overall agricultural emissions and their mitigation potential, except when considering manure that is applied to cropland soils. Indirect emissions are also considered when they relate directly to agricultural production, for example on farm energy use and emissions associated with the production of fertilizers and other agrochemicals.

The main body of the report is split into three sections. The first reviews the present level of agricultural GHG emissions, focusing particularly on N<sub>2</sub>O emissions from soils and the fertilizer applied to them, at a range of scales, and includes a discussion on the variability of field measurements and uncertainties inherent in different calculation methodologies. The CH<sub>4</sub> balance of rice production and arable farming is considered alongside issues surrounding the CO<sub>2</sub> balance of changing land use. Indirect emissions from fertilizer production and transport, agrochemical production and on-farm fuel use are also discussed. The second section reviews estimates of future agricultural GHG emission levels assuming a baseline or business-as-usual scenario, at both global and regional level, and also estimates future fertilizer requirements. The final section reviews options for mitigating these emissions, beginning with an overview of mechanisms and potential management strategies, then reviewing their global technical and economical potential, and finally provides an in-depth review of the mitigation potential of various arable cropland management options, focusing on improved nutrient management, including case studies of fertilizer use and efficiency in China and India, the two largest consumers of N fertilizer, and also covering tillage and cropping systems. Mitigation options are also reviewed with relation to rice crops specifically, since flooded cropping systems are so different from general arable conditions. In terms of the mitigation of indirect emissions, the potential for reducing emissions from fertilizer production is discussed, and the impact of reduced tillage on on-farm energy use is also reviewed.

As this report covers multiple GHGs, all emission levels have been converted into CO<sub>2</sub> equivalents according to their global warming potentials (GWP), which are taken to be 296 for N<sub>2</sub>O and 23 for CH<sub>4</sub> (IPCC, 2006), to allow easy comparison between gases.

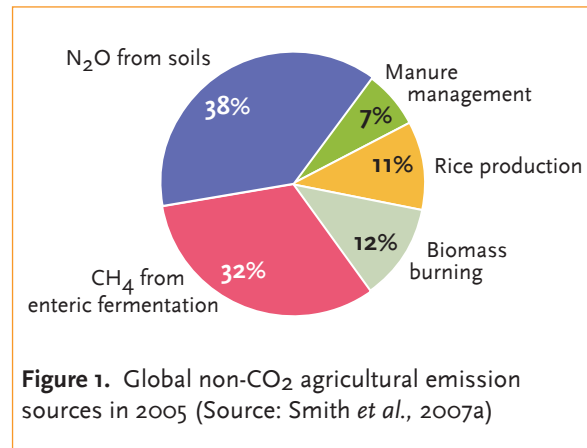
## 2. Recent and current greenhouse gas budgets

### 2.1 Overview of global and regional agricultural emissions and sources

According to the Agriculture chapter of the IPCC Fourth Assessment Report (Smith *et al.*, 2007a), approximately 40-50 % of the Earth's surface is managed for agricultural purposes and contributes 10-12 % of global greenhouse gas (GHG) emissions, around 5.1-6.1 Pg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2005. This is made up of 3.3 Pg CO<sub>2</sub>-eq yr<sup>-1</sup> from methane (CH<sub>4</sub>) and 2.8 Pg CO<sub>2</sub>-eq yr<sup>-1</sup> from nitrous oxide (N<sub>2</sub>O) emissions. Although there are large exchanges of carbon dioxide (CO<sub>2</sub>) between the atmosphere and agricultural ecosystems, emissions are thought to be roughly balanced by uptake, giving a net flux of only around 0.04 Pg CO<sub>2</sub> yr<sup>-1</sup>, less than 1 % of global anthropogenic CO<sub>2</sub> emissions (electricity and fuel use are not included in this sector) (Smith *et al.*, 2007a). Land use change is accounted for separately, but change to cultivated land is thought to contribute a further 5.9 ± 2.9 Pg CO<sub>2</sub>-eq yr<sup>-1</sup>, 6-17 % of total global GHG emissions (Bellarby *et al.*, 2008). If indirect emissions from agrochemical production and distribution and on-farm operations, including irrigation, are also included, an extra 0.4-1.6 Pg CO<sub>2</sub>-eq yr<sup>-1</sup> (0.8-3.2 %) can be attributed to agriculture, meaning that, in total, direct and indirect emissions from agricultural activity and land use change to agricultural use could contribute as much as 32.2 % of all GHG emissions (Bellarby *et al.*, 2008).

Agriculture is the main source of global non-CO<sub>2</sub> GHG emissions, contributing around 47 % of anthropogenic CH<sub>4</sub> emissions and 58 % of N<sub>2</sub>O, although there is a large degree of uncertainty around estimates for both agricultural contribution and total anthropogenic emissions. The main sources, N<sub>2</sub>O from soils and CH<sub>4</sub> from enteric fermentation, make up around 70 % of non-CO<sub>2</sub> emissions from the sector, with biomass burning, rice cultivation, and manure management, accounting for the remainder (Smith *et al.*, 2007a; see Figure 1).

The relative magnitude of emissions and sources vary greatly between different regions of the world. In Europe, agriculture contributed only 11 % of total GHG emissions in 1990, with CH<sub>4</sub> emissions from agriculture contributing 41 % of total CH<sub>4</sub> emissions, and N<sub>2</sub>O from the sector contributing 51 % of total N<sub>2</sub>O emissions (ECCP, 2001), whereas in nine African countries, agriculture contributed over 80 % of total GHG emissions in the mid 1990s (UNFCCC, 2005).



In 2005, N<sub>2</sub>O from soils (mainly associated with fertilizer and manure applications) was the main source of agricultural GHG emissions in seven out of ten world regions, while in the other three (Latin America and the Caribbean, Eastern Europe and the Caucasus and Central Asia, and OECD Pacific), CH<sub>4</sub> from enteric fermentation was the main source (US-EPA, 2006a), as these three regions had 36 % of world cattle numbers and 24 % of world sheep in 2004 (FAO, 2003). Emissions from rice cultivation and biomass burning were heavily concentrated in developing countries, with 97 % and 92 % of world totals, respectively. South and East Asia were responsible for 82 % of CH<sub>4</sub> emissions from rice cultivation as it is a dominant food source in this region, while 74 % of total emissions from biomass burning originated in Sub-Saharan Africa, and Latin America and the Caribbean. Manure management was the only source where resulting GHG emissions were higher in developed regions (52 %) than in developing regions (US-EPA, 2006a).

The balance between the large fluxes of CO<sub>2</sub> emissions and uptake in agricultural land is uncertain. A study reported in the US Environmental Protection Agency Report (US-EPA, 2006b) showed that some countries and regions have net emissions, while others have net removals of CO<sub>2</sub>. However, except for the countries of Eastern Europe, the Caucasus and Central Asia, which had an annual emission of 26 Tg CO<sub>2</sub> yr<sup>-1</sup> in 2000, all other countries showed very low estimated net fluxes, whether emissions or removals. For this reason, soil carbon fluxes are mostly discussed in terms of mitigation potential in this report (see section 4) as they are more

associated with land-use change (see section 2.6) than production from land already under agricultural management, although indirect emissions associated with fuel use and fertilizer production are covered in section 2.7. Emission levels and sources of N<sub>2</sub>O are described, focusing particularly on those associated with cropland management and fertilizer use, while CH<sub>4</sub> is mostly discussed in terms of rice production, as animal production and manure management are outside the scope of this report (see sections 2.3-2.5).

## 2.2 Recent trends (1990-present)

Globally, agricultural land use has increased by 0.8 % between 1991 and 2002, with an increase of 2.1 % in developing countries partially mitigated by a 1.5 % drop in the developed world, and now covers 5023 million hectares (Mha), 28 % of which is cropland (Smith *et al.*, 2007a). In line with this increase in activity, CH<sub>4</sub> and N<sub>2</sub>O emissions from global agriculture have increased; Vergé *et al.* (2007) report a 6.5 % increase between 1990 and 2000, while the US EPA (2006a) report a 17 % increase between 1990 and 2005 globally, with an annual average of 58 Tg CO<sub>2</sub>-eq yr<sup>-1</sup>, split roughly equally between the two gases, and with biomass burning (N<sub>2</sub>O and CH<sub>4</sub>), enteric fermentation (CH<sub>4</sub>) and soil N<sub>2</sub>O emissions together explaining 88 % of this increase.

On a regional basis, these changes represent a substantial increase in agricultural emissions from the developing world, and a decline in corresponding emissions from Europe and other, generally developed, regions. According to the US-EPA, the changes between 1990 and 2005 are composed of a 32 % increase in non-CO<sub>2</sub> emissions (equivalent to 73 Tg CO<sub>2</sub>-eq yr<sup>-1</sup>) from the five regions of Non-Annex I (developing) countries and a decrease of 12 % (equivalent to 15 Tg CO<sub>2</sub>-eq yr<sup>-1</sup>) from the other five regions (with mostly Annex I / developed countries) collectively (US-EPA, 2006a), while data from national GHG inventories shows that total agricultural GHG emissions from Annex I countries declined by 21.3 % between 1990 and 2006 (UNFCCC, 2008). This decrease was mostly due to political changes in the countries of Eastern and Central Europe, the Caucasus and Central Asia, which led to agricultural de-intensification with less inputs, and land abandonment (Smith *et al.*, 2007a). Romanovskaya (2008) reports that total agricultural emissions from Russia in 2004 were only 45 % of the 1990 levels. Vergé *et al.* (2007), who split the world into the six World Meteorological Organization Regional Associations, report that, in Africa, non-CO<sub>2</sub> agricultural production emissions (excluding biomass burning) have increased by 19 % between 1990 and 2000; in Asia by 12 %; in South America by 9 %; in North and Central America and the Caribbean

by 7 %; and in Southwest Pacific by 6%, but have decreased in Europe by 21 % over the same period.

## 2.3 N<sub>2</sub>O emissions from agricultural soils

As highlighted above, agriculture is the main source of anthropogenic N<sub>2</sub>O emissions in the world, contributing around 58 % (Smith *et al.*, 2007a), and N<sub>2</sub>O emissions from world soils amounted to 2526 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2000 (Vergé *et al.*, 2007). These emissions are produced by the microbial transformation of N in the soil, often originating from applied mineral fertilizers and manure, and can be enhanced when available N exceeds plant requirements, especially under wet conditions (Oenema *et al.*, 2005; Smith and Conen, 2004). Quantifying these emissions in order to accurately assess both their contribution to total GHG emissions and the effectiveness of mitigation strategies is, however, made difficult by the level of variation, both spatially and over time (Mosier *et al.*, 1998a). Direct N<sub>2</sub>O emissions have been shown to relate to N inputs and are therefore often calculated using an emission factor (EF), which represents the percentage of any N applied that is emitted in the form of N<sub>2</sub>O. The default EFs have large uncertainty limits, and the IPCC recently reduced its default EF from 1.25 % to 1 % as it considers a more recent review of field measurements around the world, which indicates that the initial value was too high (IPCC, 2006), while other researchers have argued that different EF values should be used for different crops, climate or soil conditions. The IPCC methodology allows country / region specific EFs to be used where data is available. This issue, and the impact using different EFs or modelling techniques can have on the quantification of emissions, are discussed further in Appendix A at the end of this report.

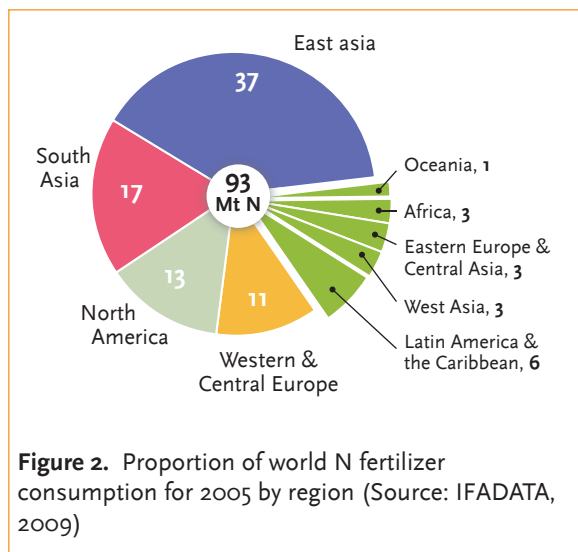
### 2.3.1 Soil N<sub>2</sub>O emissions from N fertilizer use

The availability of N fertilizers has arguably played the greatest individual role in the dramatic increase in agricultural productivity around the world since the beginning of the 20th Century. It has been estimated that, in 2008, 48% of the global population is dependent on food that would not be produced without N fertilizer inputs (Erisman *et al.*, 2008). Fertilizer use is, however, very inefficient, with a high proportion of applied N being lost to the environment. In 2005, of approximately 100 Tg N used in global agriculture, only 17 Tg N was consumed by humans as crop, dairy or meat products (UNEP, 2007), and the global N use efficiency (as measured by recovery efficiency in the first year i.e. (fertilized crop N uptake - unfertilized crop N uptake)/N applied) of crops is generally considered to be less than 50 % under most on-farm

conditions (Tilman *et al.*, 2002; Balasubramanian *et al.* 2004; Dobermann, 2007; IFA, 2007). Not all of this lost N contributes to global warming; around 40 % may be denitrified back to inert atmospheric N<sub>2</sub> (Galloway *et al.*, 2004). However, emission factors as high as 40 % (of applied N) have been reported for N<sub>2</sub>O loss from fertilizer applications (see Table A1 in Appendix A).

Data available from the International Fertilizer Industry Association (IFADATA, 2009) indicates that world N fertilizer consumption was 93 Tg N yr<sup>-1</sup> in 2005. Figure 2 shows how this was split between regions, indicating that Asia used more than 60 %. Assuming an EF of 1 % as per the IPCC default methodology (2006), this would give N<sub>2</sub>O emissions directly from N fertilizer application of 433 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2005. In comparison, Vergé *et al.* (2007) estimate N<sub>2</sub>O emissions from world N fertilizer usage at 444 Tg CO<sub>2</sub>-eq in 2000 (17.6 % of total soil N<sub>2</sub>O emissions), using FAO (2004) fertilizer consumption data and the IPCC (2000) EF of 1.25 %. This indicates the degree of variability possible in estimates even when the IPCC methodology is followed, highlighting the difficulty in quantifying both current emissions and the effectiveness of measures to mitigate them.

On a regional basis, fertilizer use contributes to total soil N<sub>2</sub>O emissions and agricultural non-CO<sub>2</sub> emissions largely in line with its level of use. Vergé *et al.* (2007) report that mineral N fertilizer applications led to emissions of 14 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in Africa in 2000, representing just 4.1 % of total soil N<sub>2</sub>O emissions and 2.2 % of total non-CO<sub>2</sub> emissions from agricultural production (excluding biomass burning). This is because African crops are generally under-fertilized; small-scale African farmers usually apply no fertilizer or rates well below the recommended levels for the maintenance of soil fertility (Batjes, 2004). In comparison, in Asia, which includes both China and India with their huge fertilizer consumption levels (see section 4.5.2 below for case studies of the impact of reducing this demand), fertilizer-induced emissions are calculated as 244 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2000, representing 22.7 % of total soil N<sub>2</sub>O emissions and 9.1 % of total non-CO<sub>2</sub> emissions from agricultural production (excluding biomass burning). South America shows a similar pattern to Africa, with emissions of 18 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2000, representing 5.2 % of soil N<sub>2</sub>O emissions and 2.4 % of non-CO<sub>2</sub> emissions from agricultural production (excluding biomass burning). The Southwest Pacific also has fertilizer induced N<sub>2</sub>O emissions of only 19 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> for 2000 but, in this case, they contribute 30.6 % of soil N<sub>2</sub>O emissions and 7.2 % of total non-CO<sub>2</sub> emissions from agricultural production (excluding biomass burning). Europe and North and Central America have similar levels of N fertilizer-induced emissions, 73 and 76 Tg CO<sub>2</sub>-eq yr<sup>-1</sup>



**Figure 2.** Proportion of world N fertilizer consumption for 2005 by region (Source: IFADATA, 2009)

for 2000, respectively, but they make a much higher contribution to total soil N<sub>2</sub>O emissions in Europe, 26.8 % vs. 17.9 % for North and Central America, despite having a fairly similar contribution to total non-CO<sub>2</sub> emissions of 11.7 % for Europe and 10.7 % for North and Central America (Vergé *et al.*, 2007).

Annex I countries (mainly developed countries or transition economies such as the Former Soviet Union) report annual emissions from fertilizer applications to the UNFCCC. Values for reported mineral N fertilizer-induced emissions range from just under 1 Gg CO<sub>2</sub>-eq yr<sup>-1</sup> for Liechtenstein to over 56 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> for the USA, which, as the third largest consumer of fertilizer N, emits almost as much directly fertilizer-induced N<sub>2</sub>O as the whole of the EU-27 (58 Tg CO<sub>2</sub>-eq yr<sup>-1</sup>). However, since the USA has a much larger cropland area (411 Mha in 2007) than the EU-27 (190 Mha in 2007, based on FAOSTAT, 2009), the average emission levels are lower on a per hectare basis. For non-Annex I countries such as China and India, emissions can instead be estimated using fertilizer consumption data from the IFA (IFADATA, 2009) and the IPCC default emission factor of 1 % (IPCC, 2006). With estimated N<sub>2</sub>O emissions from mineral N fertilizers of 148 Tg CO<sub>2</sub>-eq for 2006 (roughly 0.3 % of global total GHG emissions from all sources), China alone contributes more than the whole of Europe and the USA put together (see section 4.5.2 for further details of Chinese and Indian fertilizer use, N use efficiency and mitigation potential).

Manure application-induced N<sub>2</sub>O emissions are accounted for separately from those associated with mineral N fertilizer under the IPCC methodology. Country level emissions reported to the UNFCCC by Annex I countries are generally lower than those for mineral fertilizers due to a much smaller proportion of N being derived from manures than mineral fertilizers, ranging from under 2 Gg CO<sub>2</sub>-eq yr<sup>-1</sup> up



to almost 12 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> for the USA, and are difficult to estimate for countries which do not report them because manure applications are generally not recorded at the international level, and must first be estimated. This can be done by using animal numbers and estimates of the proportion of manure applied to crops, and its N content, but multiplying estimates means final estimates may have a very high degree of uncertainty associated with them, especially as IPCC default data on amounts of N excreted by animals may be very different to locally measured data for some countries (for example, China; Yuè and Erda, 2000). For China, manure applications have been estimated to contribute 25 % of direct soil N<sub>2</sub>O emissions, amounting to 40 Tg CO<sub>2</sub>-eq for 1990 (Yuè and Erda, 2000), although it has been suggested that the use of manure has since declined and mineral fertilizers contributed around 74 % of direct soil N<sub>2</sub>O emissions from China by the mid 1990s (Zhu and Chen, 2002), as opposed to the 54 % reported by Yuè and Erda (2000) for 1990. In India, manure applications are reported to play a more significant role in crop fertilization. Singh and Singh (2008) cite data indicating that it contributed 44 % of N inputs in 2000-01, amounting to 15.6 Tg N and producing N<sub>2</sub>O emissions of 72.6 Tg CO<sub>2</sub>-eq (assuming a 1 % EF), although the authors do note this may be an overestimate, especially as a substantial proportion of cattle excreta is used for other purposes. Bhatia *et al.* (2004) argue that manure contributes only 3 % of Indian direct soil N<sub>2</sub>O emissions, amounting to just 1.31 Tg CO<sub>2</sub>-eq for 2000-01, which agrees with Garg *et al.* (2006) in terms of percentage contribution, although Garg *et al.* (2006) estimate higher emissions overall, such that 3 % amounts to 2.79 Tg CO<sub>2</sub>-eq for 2000.

Despite being a significant source of soil N<sub>2</sub>O emissions, fertilizers do, as discussed above, make a considerable contribution towards feeding the world's increasing population (Erismann *et al.*, 2007), and when used optimally, may help to reduce total GHG emissions per unit of production by maximizing plant uptake – for example, Lammel (2009) reports that wheat and maize plants would fix around 40 % less CO<sub>2</sub> if not fertilized with N.

### 2.3.2 Soil N<sub>2</sub>O emissions per unit of agricultural output

GHG emissions are generally reported on a per area basis. However, agricultural productivity needs to increase to feed 9 billion people by 2040-2050, if we are to avoid cropland spreading into previously non-agricultural land resulting in large GHG emissions. Converting more land to crop production is likely to increase GHG emissions more than intensifying production on existing cropland (see section 4.2.7).

Therefore, lowering emissions per hectare would not be beneficial if this necessitated land conversion to cropland, so comparisons between cropping management strategies on a “per unit of production” basis is often more useful. Assessment of emissions of this basis (e.g. Williams *et al.*, 2006) is less frequent in the literature, although often done in life cycle assessment (LCA) studies (e.g. Brentrup *et al.*, 2004; Brentrup and Pallière, 2008). GHG emissions in future should be assessed on a per-unit-product basis in addition to a per-unit-area basis.

### 2.3.3 Soil N<sub>2</sub>O emissions from crop residue incorporation and N-fixing crops

Crop residues include non-harvested products and usually comprise the straw / haulms and the stubble / stover left on the field after the primary agricultural products (e.g. grain, root crops) have been removed. According to IPCC guidelines, N<sub>2</sub>O emissions from residues are calculated in the same way as N<sub>2</sub>O emissions from fertilizer inputs, by working out the level of N input from the crop and then assuming a default emission of 1 % of that input (IPCC, 2006). Input levels are determined by crop yield and N content, which varies according to crop type, and the IPCC guidelines also include an equation and a table of default factors for estimating these. For Annex I countries, crop residue incorporation is reported to contribute between 0.13 Gg CO<sub>2</sub>-eq yr<sup>-1</sup> (Iceland) and 30.36 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> (Russian Federation), with a mean average of 2067 Gg CO<sub>2</sub>-eq yr<sup>-1</sup> per country, while N-fixing crops produce 0.21-1911 Gg CO<sub>2</sub>-eq yr<sup>-1</sup>, with a mean average of 224 Gg CO<sub>2</sub>-eq yr<sup>-1</sup> per country, although this category of emissions is not reported separately for several countries, including the USA, Russia and Canada, which may be expected to have some of the highest emissions.

Given the data required, these emissions are not straightforward to estimate for other countries from readily available international databases. For China, crop residues returned to the soil were estimated by Yuè and Erda (2000) to contribute 21 % of total direct soil N<sub>2</sub>O emissions, amounting to 33 Tg CO<sub>2</sub>-eq for 1990, suggesting it is probably also one of the largest N<sub>2</sub>O emitters from crop residue incorporation. However, a lower estimate of 7.85 Tg CO<sub>2</sub>-eq (with a range of 1.57-14.13 Tg CO<sub>2</sub>-eq) is given for the same year by Xing and Yan (1999), and this study also gives a similar estimate of 8.14 (1.63-14.65) Tg CO<sub>2</sub>-eq for N<sub>2</sub>O emissions from N-fixing crops for 1990. In India, N-fixing crops added 4.1 Tg N to agricultural soils in 2000-01 (Singh and Singh, 2008), suggesting N<sub>2</sub>O emissions of 18.6 Tg CO<sub>2</sub>-eq (assuming a 1 % EF), nearly an order of magnitude higher than from any of the Annex I countries that reported this category of emissions separately. As for crop residues

are concerned, these are often used for animal feed, and only around one third are recycled (Singh and Singh, 2008). Combined with the fact that burning of crop residues is common practice in many areas of India (Sharma *et al.*, 2008), incorporation may not be a significant source of N<sub>2</sub>O emissions. Bhatia *et al.* (2004) argue that only 5 % of straw is incorporated and estimate crop residues contribute 11 % of direct soil N<sub>2</sub>O emissions, amounting to 4.09 Tg CO<sub>2</sub>-eq for 1994-95.

As with N<sub>2</sub>O emissions from fertilizer applications, however, there is considerable variation in estimated EF values, with numerous studies showing either lower or higher emissions than calculated by the IPCC default values under different conditions. For example, Velthof *et al.* (2002) reported that the total N<sub>2</sub>O emission from various types of residues ranged from 0.13 to 14.6 % of the N added with residues, and Vinther *et al.* (2004) calculated that between 1.5 and 14.1 % of total plant residue N was emitted as N<sub>2</sub>O during a five month period following incorporation. Other researchers have argued that assumptions need to be based on longer term measuring campaigns, rather than on emissions measured only during the crop growing season. Ciampitti *et al.* (2008) reported that 28 % of total N<sub>2</sub>O emissions from soybean crops occurred after harvest.

## 2.4 CH<sub>4</sub> emissions from rice cultivation

Rice paddies emit CH<sub>4</sub> when they are flooded due to the anaerobic decomposition of organic matter in the soil producing the gas, which then escapes to the atmosphere mainly through diffusive transport through the rice plants (Nouchi *et al.*, 1990), or is oxidized before reaching the surface. The level of CH<sub>4</sub> emission from any given rice paddy is related to factors that control the activity of the methane-producing (methanogens) and methane-oxidizing bacteria (methanotrophs) such as temperature, pH, soil redox potential and substrate availability, and also soil type, rice variety, water management and fertilization with organic carbon and N (see reviews by Le Mer and Roger, 2001, and Conrad, 2002). The interactions between CH<sub>4</sub> emission levels and these last three factors are discussed further in the mitigation section (4.9).

Globally, rice production is estimated to have contributed 44 % of agricultural CH<sub>4</sub> emissions in 2000, and 16 % of total non-CO<sub>2</sub> agricultural emissions on a CO<sub>2</sub>-equivalent basis (Vergé *et al.*, 2007). In 2005, 97 % of emissions from rice cultivation were from developing countries, and South and East Asia was responsible for 82 % of this, as it is a dominant food source in this region (US-EPA, 2006a). This agrees with Vergé *et al.* (2007), who estimated that Asia as a whole contributed 82 % of

CH<sub>4</sub> emissions from rice in 2000, using a global EF of 2.77 x 10<sup>-5</sup> Tg CH<sub>4</sub> km<sup>-2</sup> yr<sup>-1</sup> (calculated according to Mosier *et al.*, 1998b) to calculate total emissions from the sector from the total area under rice cultivation from the FAO (2004) database. This global emission factor corresponds to an average for the world's rice paddies, based on individual country's EFs per square kilometre, weighted for the country's contribution to the total rice paddy areas. A range of estimated CH<sub>4</sub> emission levels from rice growing countries, mostly in Asia, and world totals, are given in Table 1. Estimates of emissions from Asian countries made by Yan *et al.* (2003), were calculated using specific EFs based on reported emission levels for that country or climatically similar regions (only those estimated to emit more than 400 Gg CH<sub>4</sub> yr<sup>-1</sup> are shown in Table 1).

## 2.5 CH<sub>4</sub> budget of other croplands

In aerobic soils used for crop production, CH<sub>4</sub> production is very limited and oxidation of CH<sub>4</sub> dominates the local flux, meaning arable soils are net sinks for CH<sub>4</sub>. CH<sub>4</sub> oxidation is limited by the availability of CH<sub>4</sub>, along with other biotic and abiotic factors. As aerobic soils do not produce significant levels of CH<sub>4</sub>, the size of the sink is limited (as shown in Table 2) and estimated to amount to just 64.4 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> globally for all cultivated soils (Mosier *et al.*, 1998b). Compared to undisturbed soils, cropland soils are a weaker CH<sub>4</sub> sink (Willison *et al.*, 1995), with cultivated land consuming an average of only 46 kg CO<sub>2</sub>-eq ha<sup>-1</sup> yr<sup>-1</sup>, in comparison with temperate forests, which are thought to consume around 253 kg CO<sub>2</sub>-eq ha<sup>-1</sup> yr<sup>-1</sup> (Mosier *et al.*, 1998b). Boeckx and Van Cleemput (2001) suggest an oxidation capacity of 34.5 kg CO<sub>2</sub>-eq ha<sup>-1</sup> yr<sup>-1</sup> for European arable land, giving a sink strength of 6.3 Tg CO<sub>2</sub>-eq for the EU-15, based on land use data from 1993.

CH<sub>4</sub> oxidation in arable soils may also be limited because it is inhibited by agricultural management practices such as N fertilizer and pesticide applications. The inhibiting action of N additions is well documented (see for example, Hütsch *et al.*, 1993; Hütsch, 1996; Kravchenko *et al.*, 2002; Seghers *et al.*, 2003). It works in the short term because ammonium (NH<sub>4</sub><sup>+</sup>) interferes with the methanotrophic enzyme system (Boeckx and Van Cleemput, 1996; Tlustos *et al.*, 1998) and, in the longer term, by changing the make-up of the microbial community. Seghers *et al.* (2003) also reported that N applications in the form of mineral fertilizers had a much more negative effect than organic manure or compost applications. Various herbicides, such as atrazine, and also the insecticide methomyl, have also been shown to inhibit CH<sub>4</sub> oxidation (Topp *et al.*, 1993; Arif *et al.*, 1996; Boeckx *et al.*, 1998; Priemé and Ekelund, 2001).

Data for tropical systems, aside from rice cultivation, is rather more scarce. For example, CH<sub>4</sub> dynamics associated with agroforestry systems are poorly understood, despite complex mixtures of trees and agricultural crops (such as coffee and cacao) being widely practised in Latin America, Southeast Asia and equatorial Africa, and being considered

one of the most sustainable agricultural systems in the tropics (Albrecht and Kandji, 2003). A review by Mutuo *et al.* (2005) argues that of the few studies that have been carried out into CH<sub>4</sub> fluxes in humid tropical regions, most have focussed on short-term cropping systems or natural forests. Like temperate systems, these studies have shown that upland forests

**Table 1.** Estimated CH<sub>4</sub> emissions from rice cultivation (partially adapted from Mosier *et al.*, 1998b; and Yan *et al.*, 2003)

Country or region	CH <sub>4</sub> emission (Tg CO <sub>2</sub> -eq yr <sup>-1</sup> )	Reference
Bangladesh	17.6	ALGAS report <sup>a</sup>
	35.6	Yan <i>et al.</i> , 2003
China	299-391	Wang <i>et al.</i> , 1994
	260	Lin <i>et al.</i> , 1994
	222-291	ALGAS report <sup>a</sup>
	176	Yan <i>et al.</i> , 2003
India	55.2-138	Parashar <i>et al.</i> , 1994
	135	Yan <i>et al.</i> , 2003
	94.07 ± 27.37	Gupta <i>et al.</i> , 2009
Japan	0.46-23.92	Yagi <i>et al.</i> , 1994
	8.95	National report to UNFCCC <sup>b</sup>
	9.57	Yan <i>et al.</i> , 2003
Myanmar	30.5	ALGAS report <sup>a</sup>
	30	Yan <i>et al.</i> , 2003
Pakistan	12.1	ALGAS report <sup>a</sup>
	9.9	Yan <i>et al.</i> , 2003
Thailand	11.5-202.4	Yagi <i>et al.</i> , 1994
	48.5	National report to UNFCCC <sup>b</sup>
	40.5	ALGAS report <sup>a</sup>
	40.2	Yan <i>et al.</i> , 2003
Philippines	6.9-16.1	Neue <i>et al.</i> , 1994
	14.5	National report to UNFCCC <sup>b</sup>
	13	ALGAS report <sup>a</sup>
	12.2	Yan <i>et al.</i> , 2003
USA	5.9-7.6	US-EPA, 2009
Vietnam	29.2	ALGAS report <sup>a</sup>
	28.7	Yan <i>et al.</i> , 2003
Asia	577	Yan <i>et al.</i> , 2003
World	1380	Watson <i>et al.</i> , 1992
	584-1242	Sass, 1994
	729	Neue, 1997

<sup>a</sup>As cited by Yan *et al.* (2003); ALGAS: Asia Least-cost Greenhouse gas Abatement Strategy reports, downloaded from the website of the Asian Development Bank (ADB); <http://ntwebo3.asiandevbank.org/oes0019p.nsf/pages/sitemap>

<sup>b</sup>As cited by Yan *et al.* (2003); country communications downloaded from UNFCCC website; <http://www.unfccc.de/resource/natcom/nctable.html>

**Table 2.** CH<sub>4</sub> oxidation levels for aerobic agricultural soils (adapted from Boeckx and Van Cleemput, 2001)

Site description	Oxidation level, mean and/or range (kg CO <sub>2</sub> -eq ha <sup>-1</sup> yr <sup>-1</sup> ) <sup>a</sup>	Reference
Arable, Scotland	59 (1-153)	Dobbie <i>et al.</i> , 1996
Arable, Denmark	17 (6-24)	Dobbie <i>et al.</i> , 1996
Arable, Poland	17	Dobbie <i>et al.</i> , 1996
Winter wheat and maize	7-10	Bronson and Mosier, 1993
Arable, England	0-11	Goulding <i>et al.</i> , 1996
Arable	3-17	Mosier and Schimel, 1991
Arable, UK	25-109	Dobbie and Smith, 1994
Arable, UK	15	Willison <i>et al.</i> , 1995
Wheat, UK	69	Dobbie and Smith, 1996
Set-aside, UK	13-50	Dobbie and Smith, 1996
Abandoned farmland	24	Ambus and Christensen, 1995

<sup>a</sup>Converted from mg CH<sub>4</sub> m<sup>-2</sup> d<sup>-1</sup>, assuming the same daily rate year round.

are net sinks and that this sink strength is reduced by conversion to agriculture (Keller *et al.*, 1990; Keller and Reiners, 1994; Steudler *et al.*, 1996; Verchot *et al.*, 2000). Data from the Peruvian Amazon and lowland humid tropics in Sumatra, Indonesia, reproduced in the Mutuo *et al.* (2005) review, indicate that the CH<sub>4</sub> sink strength for forests is around 96 kg CO<sub>2</sub>-eq ha<sup>-1</sup> yr<sup>-1</sup>, while agroforestry systems can consume between 39 and 93 kg CO<sub>2</sub>-eq ha<sup>-1</sup> yr<sup>-1</sup> as CH<sub>4</sub>, and cropping systems can vary from CH<sub>4</sub> sinks of 56 kg CO<sub>2</sub>-eq ha<sup>-1</sup> yr<sup>-1</sup> for low input systems to net sources of CH<sub>4</sub> emissions of 49 kg CO<sub>2</sub>-eq ha<sup>-1</sup> yr<sup>-1</sup> for high input systems, with cassava as an example crop giving a sink strength of 48 kg CO<sub>2</sub>-eq ha<sup>-1</sup> yr<sup>-1</sup> (Tsuruta *et al.*, 2000; Palm *et al.*, 2002; scaled up from data in µg C m<sup>-2</sup> h<sup>-1</sup>, assuming a constant rate).

## 2.6 Greenhouse gas emissions from land use change

As discussed in section 2.1, the conversion of uncultivated to cultivated land is thought to contribute  $5.9 \pm 2.9$  Pg CO<sub>2</sub>-eq yr<sup>-1</sup> globally, representing 6-17 % of total GHG emissions (Bellarby *et al.*, 2008). This figure indicates both the significant contribution and the degree of uncertainty surrounding this issue.

Since 1960, agricultural area has increased from just under 4.5 to just over 4.9 billion ha in 2007 (FAOSTAT, 2009). During the last 20 years, there has been an overall increase in agricultural area from 4.86 billion ha in 1990, but showing year to year fluctuations, with the greatest area of 4.98 billion ha recorded in 2001.

The close to tripling of global food production since 1960 has largely been met through increased

food production per unit area. For example, Bruinsma (2003) suggests that 78% of the increase in crop production between 1961 and 1999 was attributable to yield increases, and 22% to expansion of harvested area, showing that whilst global agricultural area has increased only slightly, the agricultural land is managed more efficiently. Land use has therefore changed, despite smaller changes in land cover. While yield increases have outpaced increases in harvested area in most regions, the proportions vary. For example, 80% of total output growth was derived from yield increases in South Asia, compared to only 34% in sub-Saharan Africa. In industrial countries, where the amount of cultivated land has been stable or declining, increased output was derived predominantly through the development and adoption of agricultural knowledge, science and technology, which has served to increase yields and cropping intensity (van Vuuren *et al.*, 2008). The role of land use change and adoption of agricultural knowledge, science and technology has, therefore, varied greatly between regions. In some regions, particularly in Latin America, the abundance of land has slowed the introduction of new technologies (van Vuuren *et al.*, 2008).

## 2.7 Indirect emissions from crop production

### 2.7.1 Emissions from fertilizer production and distribution

Bellarby *et al.* (2008) estimate that the production of fertilizers emits between 284 and 575 Tg CO<sub>2</sub>-eq yr<sup>-1</sup>, representing 0.6-1.2 % of total global GHG emissions



**Table 3.** Total annual GHG emissions from the production, distribution and storage of mineral fertilizers, using Bellarby *et al.* (2008) methodology.

Fertilizer type	Total emissions (kg CO <sub>2</sub> -eq kg <sup>-1</sup> of fertilizer nutrient produced) <sup>a</sup>	World production in 2007 (Tg of nutrient) <sup>b</sup>	Total global GHG emission (Tg CO <sub>2</sub> -eq yr <sup>-1</sup> )
N	3.3 - 6.6	104.9	346.17 - 692.34
P	0.36 - 1.1	54.4	19.58 - 59.84
K	0.36 - 0.73	33.4	12.02 - 24.38
Total		192.7	377.77 - 776.56

<sup>a</sup>Taken from Lal (2004<sup>b</sup>)

<sup>b</sup>IFA, 2009a. N products used for industrial purposes are not included

from all sources. This is mainly due to the energy required, although nitrate production also generates N<sub>2</sub>O emissions. In total, fertilizer production uses an estimated 1.2 % of global energy demand annually, of which 94 % is used for all N fertilizer production and 87 % for ammonia production (IFA, 2009a). However, the actual energy consumed during production can vary widely as very modern fertilizer plants have the potential to efficiently use the heat produced during the reaction process and, therefore, may even be net energy producers. The most efficient ammonia production plants use natural gas as a feedstock and consume around 32 GJ per t ammonia, corresponding to 1.8 t CO<sub>2</sub> per t ammonia. The use of other feedstocks (naphtha, fuel oil, coal) requires a somewhat higher energy consumption of around 45 GJ per t ammonia, resulting in 4.1 t CO<sub>2</sub> emissions per t ammonia (IFA, 2009a). Phosphorus and potassium based fertilizers require around a tenth of energy needed for N based fertilizers (Lal, 2004b) with corresponding lower CO<sub>2</sub> emissions (IFA, 2009a).

N<sub>2</sub>O emissions from the production of nitric acid (which is used for the production of ammonium nitrate based fertilizers) are in the range of 6-8 kg N<sub>2</sub>O per t acid (1.8-2.4 t CO<sub>2</sub>-eq per t acid) for an average medium-pressure plant that has not installed N<sub>2</sub>O abatement technology (IFA, 2009a) (see section 4.10.1 for details of the mitigation potential of this technology). Holba (2009) reports that the production of ammonium nitrate fertilizer emits just under 4 t CO<sub>2</sub>-eq per t N using the best available technology in European production plants, while urea production emits around 3 t CO<sub>2</sub>-eq per t N. Due to its high GWP, N<sub>2</sub>O is thought to contribute 74 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> (26%) of the total global GHG emissions from fertilizer production, which were estimated as 283 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> by Kongshaug (1998). Table 3 shows an update of the estimate of fertilizer-associated emissions made by Bellarby *et al.* (2008), using emission levels from Lal (2004b), which include transport and storage

as well as production, and the latest data on global fertilizer production levels from the IFA.

Using updated energy requirements and correcting for proportions of different fertilizer products, the latest estimate by the IFA is 420 Tg CO<sub>2</sub>-eq for 2007; 382.8 Tg CO<sub>2</sub>-eq for production (not including ammonia etc. which is used for industrial purposes rather than fertilizers) plus 37.2 Tg CO<sub>2</sub>-eq for transport and distribution (IFA, 2009a). This means that globally, emissions from fertilizer production, distribution, and storage account for around 0.8 % of total global GHG emissions. The potential to mitigate these emissions by further implementation of best practice technology is discussed in section 4.10.1.

### 2.7.2 Emissions associated with other agrochemicals

The production of crop protection products is estimated to account for 3-140 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> by Bellarby *et al.* (2008), based on per km<sup>2</sup> emissions of 220-9220 kg CO<sub>2</sub>-eq, calculated using emissions per kg from Lal (2004c) and an application rate of 0.5-2 kg ha<sup>-1</sup> based on Clemens *et al.* (1995), multiplied by a total cropland area of 15.41 million km<sup>2</sup> (in 2003; FAOSTAT, 2007). A recent study into herbicide application, energy efficiency and CO<sub>2</sub> emissions from cereal cropping, reported that the application of herbicides increases CO<sub>2</sub> emissions by 4.4 %, because of the energy required for the production and application of the herbicides, as well as the greater amount of energy required for harvesting caused by higher yields, but that overall, herbicide application increased energy efficiency, with emissions per unit of grain equivalent produced reduced by 36.4 % (Deike *et al.*, 2008). Similarly, Berry *et al.* (2008) report that fungicide application to UK wheat crops produces CO<sub>2</sub> emissions of 0.06 Tg CO<sub>2</sub>-eq yr<sup>-1</sup>, but that emissions per unit of grain produced are reduced, such that an extra 0.93 Tg CO<sub>2</sub>-eq would be produced

per year to maintain the same yield without fungicide applications, because reduced yields would mean greater areas would need to be cultivated. A similar argument can be made if yields are reduced due to under-fertilization, since additional land would be required to replace the lost product. The magnitude of these “displaced emissions” from reduced yield depends upon where the new cropland is sourced (Carlton *et al.*, 2009).

### 2.7.3 On-farm energy use emissions

Farm operations such as tillage, seeding, application of agrochemicals and harvesting are variable across the globe, with emissions ranging between 59 and 257 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> (Bellarby *et al.*, 2008; see Table 4.). These calculations agree with Frye (1984), who reported that tillage and harvest represent the greatest proportion of fuel consumption within intensive conventional agricultural systems. Irrigation is estimated to emit 3.44-4.44 t CO<sub>2</sub>-eq km<sup>-2</sup> yr<sup>-1</sup> (Lal, 2004b), leading Bellarby *et al.* (2008) to estimate

global emissions of between 50 and 680 Tg CO<sub>2</sub>-eq yr<sup>-1</sup>, although this is based on total global cropland areas and is therefore likely to be an overestimate, given that not all cropland is irrigated. According to the FAO AQUASTAT database, around 20 % of global croplands are irrigated, so emissions from irrigation may actually be more in the range of 10-136 Tg CO<sub>2</sub>-eq yr<sup>-1</sup>.

In terms of global energy use, agriculture consumes a relatively low proportion of fossil fuels (Saurbeck, 2001); even high intensity farming in well developed industrialized countries only consumes about 3-4.5 % of the countries’ total energy budget, if only primary farm production and not secondary food industry activities (i.e. processing, preserving and transport) are taken into account (CAST, 1992; Enquete Commission, 1995). According to Nelson *et al.* (2009), energy use for crop production in the USA accounted for CO<sub>2</sub> emissions of 78.8-85.1 Tg yr<sup>-1</sup> between 1990 and 2004. See section 4.10.2 for discussion on the mitigation of this class of emissions.

**Table 4.** GHG emissions from the use of farm machinery for different operations (adapted from Bellarby *et al.*, 2008)

Operation type	Emission level (kg CO <sub>2</sub> -eq ha <sup>-1</sup> ) <sup>a</sup>	Estimated global emissions (Tg CO <sub>2</sub> -eq yr <sup>-1</sup> ) <sup>b</sup>
Tillage	4.40-73.60	7-113
Application of agrochemicals	1.80-37.00	3-57
Drilling or seeding	8.10-14.30	15-22
Combine harvesting	22.10-42.10	34-65
Total		59-257

<sup>a</sup>Calculated from data in Lal (2004c)

<sup>b</sup>Calculated using the area under permanent crops and arable land (15.41 million km<sup>2</sup>) in 2003 (FAOSTAT, 2007).

### 3. Future baseline emissions

**B**aseline predictions are based on ‘business-as-usual’ scenarios that assume current trends in agricultural production (see below) will continue, such that the world will continue to develop much as it does today, and that no new policies will be introduced, either in response to these developments or to reduce GHG emissions. They are used, therefore, to help assess the possible impact of individual policy changes or mitigation options on future emission levels.

#### 3.1 General trends in future agriculture

The IPCC Working Group III (contribution to the IPCC Fourth Assessment Report, Smith *et al.*, 2007a) considered the main trends in the agricultural sector with implications for future GHG emissions to be:

##### **Land use and productivity**

Land productivity will continue to increase, albeit at a declining rate, as further technological progress gives declining returns, and greater use is made of marginal land with lower productivity. This increases the risk of soil erosion and degradation, with highly uncertain consequences for CO<sub>2</sub> emissions (Lal, 2004a; Van Oost *et al.*, 2004).

##### **Tillage practices**

Conservation tillage and zero-tillage are increasingly being adopted (accounting for 3.5 % of arable land by 1999, according to the FAO (2001)), reducing energy usage from reduced on-farm fuel use, despite higher embedded energy in the increased herbicide use, and often increasing carbon storage in soils. However, assessing the effect of this on the GHG balance is difficult, especially as they are often combined with periodic tillage.

##### **Additional inputs**

Irrigation and fertilizer use increases will be required to support increasing productivity, in turn increasing energy demands (for moving water and manufacturing fertilizer; Schlesinger, 1999). This may lead to increased GHG emissions on a per-area basis (Mosier, 2001), although this depends on the efficiency of water and fertilizer use, and lower GHG emissions per-unit-product.

##### **Livestock**

Growing demand for meat is likely to induce changes in land use, particularly increases in grassland, and also increased demand for animal feeds (e.g. cereals), with associated increases in CO<sub>2</sub> emissions. Larger herds of beef cattle will increase emissions of CH<sub>4</sub> and N<sub>2</sub>O, although this may be partially mitigated by the use of intensive systems (with lower emissions per unit product), which are expected to increase at a faster rate than grazing-based systems. Increases in manure production will also increase GHG emissions.

##### **Transport and energy**

Changes in policies such as subsidies, and regional patterns of production and demand are causing an increase in international trade of agricultural products. This is expected to increase CO<sub>2</sub> emissions, due to greater use of energy for transportation. However, there is also an emerging trend for greater use of agricultural products (e.g. bioplastics, biofuels and biomass for energy) as substitutes for fossil fuel-based products, which has the potential to reduce GHG emissions in the future.

#### 3.2 Overview of baseline predictions

When the IPCC Fourth Assessment Report (Smith *et al.*, 2007a) was prepared, no baseline agricultural non-CO<sub>2</sub> GHG emission estimates for the year 2030 had been published. Instead, the report includes an estimate based on the US-EPA (2006a) prediction that aggregate emissions will increase by ~13 % during the decades 2000-2010 and 2010-2020. Assuming similar rates of increase (10-15 %) for 2020-2030, agricultural emissions might be expected to rise to 8000–8400 Tg CO<sub>2</sub>-eq yr<sup>-1</sup>, with a mean of 8300 Tg CO<sub>2</sub>-eq by 2030. Vergé *et al.* (2007) are in broad agreement with this, predicting that non-CO<sub>2</sub> emissions from agricultural production (excluding those from biomass burning) will reach 8189 Tg CO<sub>2</sub>-eq by 2030. These increases will be driven by increases in production to support population growth in areas such as South and East Asia, Latin America and Africa, and increased affluence leading to changes in dietary preferences and increased commercialization of production, with associated increases in fertilizer usage and animal rearing (Smith *et al.*, 2007a). Rosegrant *et al.* (2001)

project that an additional 500 Mha will be converted to agriculture during 1997-2020, mostly in Latin America and Sub-Saharan Africa, and that there will be a 57 % increase in global meat demand over the same period, mostly in South and Southeast Asia, and Sub-Saharan Africa.

Using a baseline scenario taken from a study published by the Netherlands Environmental Assessment Agency (MNP, 2006), which projects the world developing over the next decades very much as it does today, without anticipating deliberate interventions or responses to the projected developments, and with no implementation of policies for emission reduction, Verburg *et al.* (2008) report modelled GHG emissions up to 2050. They predict that CO<sub>2</sub> emissions from land use (agriculture including land-use change) will show a net increase of approximately 18 Tg yr<sup>-1</sup> between 2000 and 2015, with the strongest increases in Asia, Africa and the Americas, while a decrease in emissions is found in China. Between 2015 and 2030, there will be a global net decrease in CO<sub>2</sub> emissions of approximately 30 Tg yr<sup>-1</sup>, with largest declines in Latin America, Asia, and the OECD Pacific, and increased emissions in Europe, Russia, Africa and North America. By 2030, the modelling study estimates that CH<sub>4</sub> emissions from ruminants and manure will reach around 3200 Tg CO<sub>2</sub>-eq yr<sup>-1</sup>, and N<sub>2</sub>O emissions (soil including fertilizer, and manure emissions) will reach around 1300 Tg CO<sub>2</sub>-eq yr<sup>-1</sup>, with both gases showing the strongest increases in Africa.

Focusing on emissions related to crop production, N<sub>2</sub>O from agricultural soils, the largest contributor to agricultural GHG emissions, is predicted to increase by 47 % (compared to 1990 levels) to 2937 Tg CO<sub>2</sub>-eq by 2020 by the US-EPA (2006a), while the FAO estimates a 35-60 % increase in total agricultural N<sub>2</sub>O emissions (including those from manure) by 2030 (FAO, 2003). CH<sub>4</sub> emissions from rice cultivation are predicted to increase from 601 Tg CO<sub>2</sub>-eq in 1990 to 776 Tg CO<sub>2</sub>-eq by 2020 (US-EPA, 2006a), assuming a sustained increase in irrigated production. The FAO, however, forecasts that the global area under rice cultivation will grow by just 4.5 % to 2030 (FAO, 2003) and, therefore, emissions may increase little from this source, especially if rice is grown under continuously flooded conditions, or if new lower CH<sub>4</sub> emitting rice cultivars are developed and adopted (Wang *et al.*, 1997). Future CO<sub>2</sub> emission levels from agriculture are uncertain, but most likely to decrease or remain at low levels, for example due to increased adoption of conservation tillage practices (FAO, 2001). However, if emissions from land use change are included, CO<sub>2</sub> emissions may increase; Verburg *et al.* (2008) suggest a strong increase of CO<sub>2</sub> emissions up to 2020, caused by land clearing of natural vegetation for agricultural

land use in Africa, Latin America, Southeast Asia including Indonesia, and South Asia including India.

### 3.3 Regional trends

The IPCC Fourth Assessment (Smith *et al.*, 2007a) considers that the highest projected growth in emissions is for the Middle East and North Africa, and Sub-Saharan Africa, with a combined 95% increase in the period 1990 to 2020 (US-EPA, 2006a). Sub-Saharan Africa is the one region of the world where per-capita food production is currently either in decline, or roughly constant at a level that is not adequate (Scholes and Biggs, 2004), in part due to low and declining soil fertility (Sanchez, 2002) and inadequate fertilizer inputs. The slowly rising wealth of urban populations is likely to increase demand for livestock products, resulting in intensification of agriculture and expansion into areas which are currently largely unexploited, particularly in Southern and Central Africa (including Angola, Zambia, DRC, Mozambique and Tanzania), and a consequent increase in GHG emissions. However, Vergé *et al.* (2007) consider that, in Africa, the AIDS epidemic may impact on this projected growth as, according to FAO estimates, AIDS has killed some seven million of Africa's agricultural workers and could result in 16 million more deaths by 2020, removing nearly a quarter of African agricultural workers from the labour pool within 20 years (CGIAR, 2001). On the other hand, since current agricultural practices in Africa are extremely labour-intensive, a shrinking workforce means that there is a need for increased productivity per worker (as well as per hectare) unless workers can be lured back to agriculture through improved rural livelihoods. Verburg *et al.* (2008) estimate that African CH<sub>4</sub> emissions from ruminants and manure will increase by approximately 10 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2000-2015, and by approximately 12 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2015-2030, while N<sub>2</sub>O emissions from soils and manure from the region will increase by approximately 3 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2000-2015, and by approximately 4 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2015-2030.

In East Asia, a large increase in GHG emissions from animal sources is projected. Between 1961 and 2004, total production of meat in Asian developing countries increased more than 12 times and milk production by more than four times (FAOSTAT, 2006). Since the per-capita consumption of meat and milk is still much lower in these countries in comparison with developed countries, these trends are expected to continue for a relatively long time, and accordingly, US-EPA (2006a) forecast increases of 153% and 86% in emissions from enteric fermentation and manure management, respectively, from 1990 to 2020. In South Asia, emission increases are mostly due to

expanding use of N fertilizers and manure to meet demands for food, resulting from rapid population growth. Verburg *et al.* (2008) estimate that CH<sub>4</sub> emissions (from ruminants and manure) from South Asia will increase by ~8 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2000-2015 and ~4 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2015-2030, while those from Southeast Asia will increase by ~2 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2000-2015, and ~1 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2015-2030, and those from China will increase by ~4 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2000-2015, and by ~2 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2015-2030. Emissions of N<sub>2</sub>O from soils and manure, meanwhile, are forecast to increase by a total of ~4 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2000-2015 for South and Southeast Asia combined, and by ~1 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> for China over the same period, while in 2015-2030, China shows a decline of ~1 Tg CO<sub>2</sub>-eq yr<sup>-1</sup>, with South and Southeast Asia together still increasing by a total of ~2 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> (Verburg *et al.*, 2008).

In Latin America and the Caribbean, agricultural products are the main source of exports. Significant changes in land use and management have occurred, especially forest conversion to cropland and grassland, resulting in increased GHG emissions from soils (CO<sub>2</sub> and N<sub>2</sub>O). Between 1961 and 2004, the cattle population increased linearly from 176 to 379 million heads, partly offset by a decrease in the sheep numbers from 125 to 80 million heads. All other livestock categories have increased in the order of 30 to 600 % since 1961, and cropland areas, including rice and soybean, and the use of N fertilizers have also shown dramatic increases (FAOSTAT, 2006). Another major trend in the region is the increased adoption of no-till agriculture, particularly in Brazil, Argentina, Paraguay and Uruguay. This is practised on ~30 Mha every year in the region, although it is not known how much of this is under permanent no-till management (Smith *et al.*, 2007a). Verburg *et al.* (2008) estimate that agricultural CH<sub>4</sub> emissions from Latin America (including Brazil) will increase by ~8 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2000-2015 and by ~6 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2015-2030, while agricultural N<sub>2</sub>O emissions from the same region will increase by ~1.5 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2000-2015 and by ~0.5 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2015-2030.

In the countries of Central and Eastern Europe and Central Asia, agricultural production is currently about 60-80% of that in 1990, but is expected to grow by 15-40 % above 2001 levels by 2010, as these countries increase in wealth. Arable land area is forecast to increase by 10-14 % for the whole of Russia due to agricultural expansion. The widespread move to more intensive management could result in a 2 to 2.5-fold rise in grain and fodder yields, with a consequent reduction in arable land, but possible increase N fertilizer use. Decreases in fertilizer N use since 1990 have led to a significant reduction in N<sub>2</sub>O emissions but, under favourable economic

conditions, applications are expected to rise again, although they are unlikely to reach pre-1990 levels in the near future (Smith *et al.*, 2007a). US-EPA (2006a) projects a 32% increase in N<sub>2</sub>O emissions from soils in these two regions between 2005 and 2020, which is equivalent to an average increase rate of 3.5 Tg CO<sub>2</sub>-eq yr<sup>-1</sup>. In comparison, Verburg *et al.* (2008) estimate that agricultural N<sub>2</sub>O emissions from Eastern Europe, Central Asia, Russia and the Caucasus will increase by ~2 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2000-2015, and by ~1 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2015-2030. Emissions of CH<sub>4</sub> for the region are predicted to increase by ~1 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2000-2015, and by ~0.5 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2015-2030 (Verburg *et al.*, 2008).

The only developed regions showing a consistent increase in GHG emissions in the agricultural sector are North America and developed countries of the Pacific, with increases of 18 % and 21 %, respectively, between 1990 and 2020. In both cases, this trend is largely driven by CH<sub>4</sub> and N<sub>2</sub>O emissions from manure management and soils. In Oceania, N fertilizer use, although still very low in international terms, has increased sharply with a five fold increase since 1990 in New Zealand and a 2.5 fold increase in Australia over the same period (Smith *et al.*, 2007a). The practice of burning sugar cane residues in Australia may contribute to this as it reduces nutrient returns to the soil (Vergé *et al.*, 2007). In contrast, North American N fertilizer use levels have remained stable for around 25 years and here the main driver for increasing emissions is management of manure from cattle, poultry and swine production, and manure application to soils. The US-EPA estimates that CH<sub>4</sub> and N<sub>2</sub>O emissions associated with manure management have increased by almost 40 % (added together, on a CO<sub>2</sub>-eq basis) between 1990 and 2007 (US-EPA, 2009). In both regions, conservation policies have reduced CO<sub>2</sub> emissions from land conversion; land clearing in Australia has declined by 60 % since 1990 with vegetation management policies restricting further clearing, while, in North America, some marginal croplands have been returned to woodland or grassland (Smith *et al.*, 2007a). Verburg *et al.* (2008) provide more conservative estimates which do not include increases from 1990 to 2000. They estimate that CH<sub>4</sub> emissions from North America will increase by ~2 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2000-2015 and by <1 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2015-2030, while N<sub>2</sub>O emissions from the region will increase by <1 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2000-2015 and decrease by <0.5 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> in 2015-2030. For Oceania, even smaller increases in CH<sub>4</sub> are predicted and changes in N<sub>2</sub>O emissions are negligible.

Western Europe is the only region where, according to the US-EPA (2006a), GHG emissions from agriculture are projected to decrease by 2020. This is



**Table 5.** Projected non-CO<sub>2</sub> emissions (Tg CO<sub>2</sub>-eq yr<sup>-1</sup>) from agricultural production (excluding biomass burning) for 2015 and 2030 by region (adapted from Vergé *et al.*, 2007).

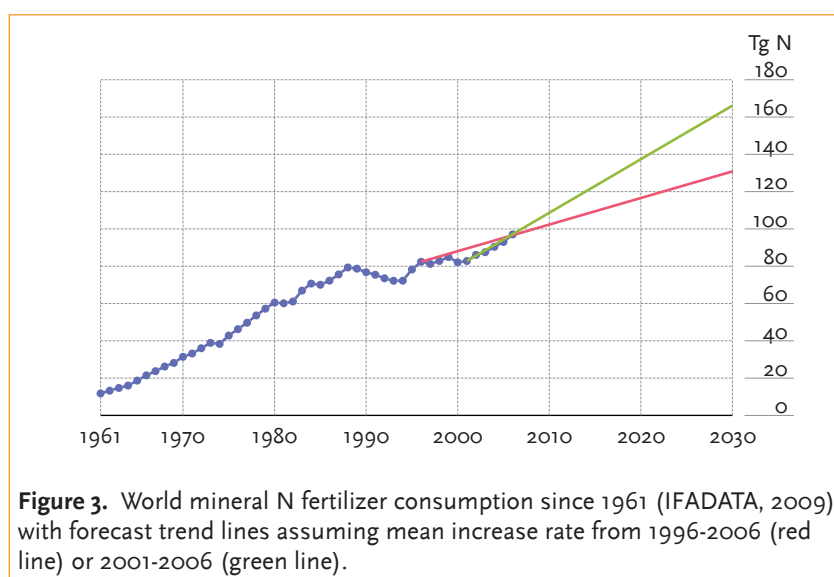
Region	2015		2030	
	emissions	change 2015-2000	emissions	change 2030-2000
Africa	796	27 %	1422	127 %
Asia	3203	20 %	3788	42 %
S. America	966	28 %	1207	59 %
N. and Central America	789	11 %	877	23 %
SW Pacific	296	12 %	329	25 %
Europe	594	-5 %	566	-10 %
World total	6644	17 %	8189	45 %

associated with the adoption of a number of climate-specific and other environmental policies in the European Union (EU), as well as economic constraints on agriculture. For example, successive reforms of the Common Agricultural Policy (CAP) since 1992 have contributed to GHG reduction, and the 2003 EU CAP reform is expected to continue this trend, mainly through reduction of animal numbers (Binfield *et al.*, 2006). Verburg *et al.* (2008) predict a decline in CH<sub>4</sub> emissions, but these are outweighed by an increase in N<sub>2</sub>O emissions. However, the baseline scenario in this study does not include the implementation of any new agricultural or environmental policies, and Central Europe is included in the same category as Western Europe so the region is not directly comparable either.

Table 5 shows the projected non-CO<sub>2</sub> agricultural emission levels (excluding biomass burning) for the six World Meteorological Organization Regional Associations for 2015 and 2030, calculated by Vergé *et al.* (2007) based on projected food requirements, given population evolution according to the World Resource Institute (2004) and FAO (2004). The latest inventory from the US-EPA suggests that the increases in emissions for this region may have been over-estimated, as non-CO<sub>2</sub> emissions from US agriculture have only increased by 7.5 % on a CO<sub>2</sub>-eq basis between 1990 and 2007 (US-EPA, 2009).

### 3.4 Future fertilizer use and associated emissions

Erisman *et al.* (2008) used the IPCC Special Report on Emission Scenarios (SRES) (Nakicenovic *et al.*, 2000) to predict future fertilizer demand depending on economic, demographic and technological developments, specifically population growth, N use efficiency, biofuel production, increased meat consumption and other dietary changes. They estimate N fertilizer demand will be between approximately 100 and 140 Tg N by 2030. In comparison, the FAO (2000) predict baseline fertilizer consumption by 2030 of around 135 Tg N and also give an estimate of 95 Tg N based on a scenario of improved nutrient use efficiency, and Tilman *et al.* (2001) forecast usage of 135 Tg N by 2020, increasing to 236 Tg N by 2050, suggesting a level of around 170 Tg N by 2030 if a linear increase is assumed. Work done by

**Figure 3.** World mineral N fertilizer consumption since 1961 (IFADATA, 2009), with forecast trend lines assuming mean increase rate from 1996-2006 (red line) or 2001-2006 (green line).

the International Energy Agency (IEA) suggests a baseline consumption of 121 Tg N yr<sup>-1</sup> by 2020 and 129 Tg N yr<sup>-1</sup> by 2030 but suggests that policy changes promoting the use of biofuels could increase this to 128 Tg N yr<sup>-1</sup> by 2020 and 149 Tg N yr<sup>-1</sup> by 2030 (Gielen, 2006). Figure 3 illustrates the increase in world N fertilizer consumption from 1961 to 2006 according to the IFA, indicating that the decline in N applications from the late 1980s due to the collapse of the Former Soviet Union only lasted until 1994, and more recently levels have increased steadily. Assuming the mean increase rate of either 1996-2006 (red line on Figure 3) or 2001-2006 (green line), which illustrates the sudden increase in demand partly caused by interest in biofuels, gives a business-as-usual baseline consumption of 118-137 Tg N yr<sup>-1</sup> by 2020, and 132-166 Tg N yr<sup>-1</sup> by 2030.

Currently, the IFA forecasts that N fertilizer consumption will reach 102 Tg N yr<sup>-1</sup> in 2009/10

(Heffer and Prud'homme, 2009) in light of current economic problems, and 111.1 Tg N yr<sup>-1</sup> by 2013/14, which suggests growth may be nearer the lower estimate, at least for the next few years. Even so, this lowest business-as-usual projection of mineral N fertilizer consumption of 118 Tg N yr<sup>-1</sup> by 2020 may produce soil N<sub>2</sub>O emissions of around 550 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> (assuming a 1 % EF), which is an increase of 24 % over 20 years using the estimate for 2000 made by Vergé *et al* (2007). This will also increase indirect emissions associated with fertilizer production, with the increase being highly dependent on the proportion of production plants using the best available technology (BAT). For example, new ammonia production plants using BAT today produce less than 40 % of the GHG emissions of the average plant using coal, naphtha or oil as a feedstock (IFA, 2009a) (see sections 2.7.1 and 4.10.1).

## 4. Mitigation potential

### 4.1 Mitigation options and mechanisms

Agricultural GHG fluxes are produced by complex and heterogeneous mechanisms, but the active management of agricultural systems offers possibilities for mitigation, many using current technologies which could be implemented immediately. These mitigation practices include cropland management, grazing land management, management of agricultural organic soils, restoration of degraded lands, livestock management, manure/biosolid management, and bioenergy production. This report will focus on practices related to cropland, and particularly to nutrient management.

Mitigation opportunities can be based on reducing emissions, enhancing removals or storage, or avoiding or displacing emissions (Smith *et al.*, 2007b). This third category includes using biofuels instead of fossil fuels and avoiding bringing new areas of land under agricultural cultivation. Practices that reduce emissions seek to better manage the flows of carbon and N within the agricultural ecosystem. For example, practices that increase crop N use efficiency often reduce N<sub>2</sub>O emissions (Bouwman, 2001). Practices that enhance removals or storage act to sequester carbon or build carbon sinks, by increasing the photosynthetic input of carbon and/or slowing the return of stored carbon to CO<sub>2</sub> via respiration, fire or erosion. Agricultural ecosystems contain large carbon reserves (IPCC, 2001a), mostly in the form of soil organic matter. Historically, these systems have lost more than 50 Pg C (Paustian *et al.*, 1998; Lal, 1999, 2004a), but some of this can be recovered through improved management. Vegetation can also store significant amounts of carbon in agro-forestry systems or other perennial plantings on agricultural lands (Albrecht and Kandji, 2003). Agricultural lands have been shown to remove CH<sub>4</sub> from the atmosphere by oxidation processes, although forests remove more (Tate *et al.*, 2006), and this effect is small compared to other GHG fluxes (Smith and Conen, 2004).

Many practices have been suggested to mitigate emissions, and they often work by more than one mechanism, and affect more than one gas, sometimes in opposite ways. Therefore, it is important to consider the net benefit, which depends on the combined effects on all gases (Robertson and Grace, 2004; Schils *et al.*, 2005; Koga *et al.*, 2006). Temporal patterns of emissions also need to be taken into consideration, as

these may vary between practices or for different gases for a given practice; in some cases, emissions can be reduced indefinitely, while other reductions may be temporary (Marland *et al.*, 2003a; Six *et al.*, 2004). Finally, it must be considered whether practices affect radiative forcing through other mechanisms; for example, aerosols or albedo (Marland *et al.*, 2003b; Andreae *et al.*, 2005).

### 4.2 Overview of cropland management mitigation practices

Croplands have a high potential for mitigating net GHG emissions as they are often intensively managed. Practices that could be implemented fall into the following partially overlapping categories:

#### 4.2.1 Agronomy

Improved agronomic practices increase yields and generate higher inputs of carbon residue and can, therefore, increase soil carbon storage (Follett, 2001). Examples include using improved crop varieties, extending crop rotations, especially those with perennial crops that allocate more carbon below ground, and avoiding or reducing fallow periods (West and Post, 2002; Lal, 2003, 2004a; Freibauer *et al.*, 2004; Smith, 2004a, b). Adding more nutrients, when the soil is poor, can also promote soil carbon gains (Alvarez, 2005), but the benefits may be offset by higher N<sub>2</sub>O emissions from soils and CO<sub>2</sub> from fertilizer manufacture (Schlesinger, 1999; Pérez-Ramírez *et al.*, 2003; Robertson, 2004; Gregorich *et al.*, 2005).

Emissions per hectare can also be reduced by using cropping systems that are less reliant on fertilizers, pesticides and other inputs, thereby reducing GHG costs from their production as well (Paustian *et al.*, 2004). The use of rotations with legume crops is an important example of this (West and Post, 2002; Izaurralde *et al.*, 2001), as the biologically fixed N reduces the need for external N inputs, although it can still be a source of N<sub>2</sub>O emissions (Rochette and Janzen, 2005; Parkin and Kaspar, 2006; Ciampitti *et al.*, 2008). The GHG balance of legumes may not always be as straightforward as this suggests. Soybeans in rotation with maize have been shown to reduce soil carbon storage in comparison with continuous maize under some management practices (Adviento-Borbe *et al.*, 2007). It has been suggested that the greater



amount of N available to maize crops planted after soybeans is not due to a net input of N from the atmosphere, but instead the result of mineralization of soybean residues and enhanced mineralization of soil organic matter because soybean residues have a lower C:N ratio than maize residues (Salvagiotti *et al.*, 2008). Reducing emissions per unit area may not reduce total GHG emissions from crop production if the measures reduce productivity and require extra land to be taken into agricultural production (Carlton *et al.*, 2009).

Another group of agronomic practices are those that provide temporary vegetative cover between successive agricultural crops, or between rows of tree or vine crops, known as ‘catch’ or ‘cover’ crops. These add carbon to soils (Arrouays *et al.*, 2002; Barthès *et al.*, 2004; Freibauer *et al.*, 2004; Ogle *et al.*, 2005) and may also reduce N<sub>2</sub>O emissions by extracting plant-available N unused by the preceding crop (Velthof and Kuikman, 2000). The mitigation potential of various measures within this category is discussed further in section 4.8 below.

#### 4.2.2 Nutrient management

Nitrogen applied in the form of fertilizers, manures, biosolids and other N sources is not always used efficiently by crops (Galloway *et al.*, 2003; Cassman *et al.*, 2003), and the surplus is particularly susceptible to emission of N<sub>2</sub>O (McSwiney and Robertson, 2005). As a result, improving N use efficiency (NUE) can reduce N<sub>2</sub>O emissions, both on- and off-site, by also reducing leaching and volatile losses, and indirectly reduce GHG emissions from N fertilizer manufacture (Schlesinger, 1999). Practices that improve NUE include adjusting application rates based on precise estimation of crop needs; using slow- or controlled-release fertilizers or nitrification inhibitors (which slow the microbial processes leading to N<sub>2</sub>O formation); improving timing by applying N when it is least susceptible to loss, often just prior to plant uptake; making N more accessible to crop roots by placing it more precisely into the soil; or avoiding N applications in excess of immediate plant requirements (Robertson, 2004; Dalal *et al.*, 2003; Paustian *et al.*, 2004; Cole *et al.*, 1997; Monteny *et al.*, 2006). In many countries, the fertilizer industry is advancing the principles of sound nutrient management via an approach termed “4R Nutrient Stewardship” (IFA, 2009b), promoting fertilizer best management practices. Measures within this category are discussed further below and their mitigation potential is examined for several case studies (see sections 4.5 and 4.6).

#### 4.2.3 Tillage/residue management

Advances in weed control methods and farm machinery now allow many crops to be grown with minimal tillage (reduced tillage) or without tillage (no-

till), and these practices are increasingly being used around the world (e.g. Cerri *et al.*, 2004). Because soil disturbance tends to stimulate carbon losses through enhanced decomposition and erosion (Madari *et al.*, 2005), reduced- or no-till agriculture often results in soil carbon gain, but this is not always the case (West and Post, 2002; Ogle *et al.*, 2005; Gregorich *et al.*, 2005; Alvarez 2005). Some researchers (e.g. Baker *et al.*, 2007; Baker and Griffis, 2005) have argued that sampling protocols have biased the results, and that different tillage regimes affect the depth distribution of SOC, with conservation tillage leading the higher levels near the surface and conventional tillage giving higher C levels deeper in the soil profile (Carter, 2005; Dolan *et al.*, 2006), suggesting that some studies may report a redistribution rather than an increase in soil C (Powlson and Jenkinson, 1981; Machado *et al.*, 2003). Nevertheless, when such studies are excluded from analysis, most studies do show modest increases in SOC under zero/reduced tillage (Ogle *et al.*, 2005).

Adopting reduced- or no-till may also affect N<sub>2</sub>O emissions but the net effects, which seem to depend on soil and climatic conditions, are inconsistent and not well quantified globally (Cassman *et al.*, 2003; Smith and Conen, 2004; Helgason *et al.*, 2005; Li *et al.*, 2005). In some areas, reduced tillage promotes N<sub>2</sub>O emissions, while elsewhere it may reduce emissions or have no measurable influence (Marland *et al.*, 2001). No-tillage systems can, however, reduce CO<sub>2</sub> emissions from energy use (Marland *et al.*, 2003b; Koga *et al.*, 2006). Retaining crop residues also tends to increase soil C because they are the precursors for soil organic matter, which is the main C store in soil. Avoiding the burning of residues, e.g. mechanizing sugar cane harvesting and eliminating the need for pre-harvest burning (Cerri *et al.*, 2004) also avoids emissions of aerosols and GHGs generated from fire, although CO<sub>2</sub> emissions from fuel use may increase. The effectiveness of these measures as mitigation practices in terms of their impacts on both C storage and N<sub>2</sub>O emissions are examined further below (see section 4.7).

#### 4.2.4 Water management

Around 18% of global croplands now receive supplementary water through irrigation (Millennium Ecosystem Assessment, 2005). Expanding this area (where water reserves allow) or using more effective irrigation measures can enhance C storage in soils through higher yields and residue returns (Follett, 2001; Lal, 2004a). However, some of these gains may be offset by CO<sub>2</sub> from energy used to deliver the water (Schlesinger 1999; Mosier *et al.*, 2005) or from N<sub>2</sub>O emissions from higher moisture and fertilizer N inputs (Liebig *et al.*, 2005). In humid regions, drainage can promote crop productivity (and hence soil C)

and may also suppress N<sub>2</sub>O emissions by improving aeration (Monteny *et al.*, 2006). Any N lost through drainage, however, may be susceptible to loss as N<sub>2</sub>O (Reay *et al.*, 2003). Modelled estimates of the global mitigation potential of this measure are given in Table 6.

#### 4.2.5 Rice management

Cultivated wetland rice soils emit significant quantities of CH<sub>4</sub> (Yan *et al.*, 2003), but emissions during the growing season can be reduced by various practices (Yagi *et al.*, 1997; Wassmann *et al.*, 2000; Aulakh *et al.*, 2001). For example, draining the rice once or several times during the growing season reduces CH<sub>4</sub> emissions (Smith and Conen, 2004; Yan *et al.*, 2003; Khalil and Shearer, 2006). This benefit may be partly offset by increased N<sub>2</sub>O emissions though (Akiyama *et al.* 2005). Rice cultivars with low exudation rates could offer an important methane mitigation option (Aulakh *et al.*, 2001; Hou *et al.*, 2000). In the off-rice season, CH<sub>4</sub> emissions can be reduced by improved water management, especially by keeping the soil as dry as possible and avoiding water logging (Cai *et al.*, 2000, 2003; Kang *et al.*, 2002; Xu *et al.*, 2003). Adjusting the timing of organic residue additions (e.g. incorporating organic materials in the dry period rather than in flooded periods; Xu *et al.*, 2000; Cai and Xu, 2004), and composting the residues before incorporation can also reduce CH<sub>4</sub> emissions, and there is potential for producing biogas for use as fuel for energy production (Wang and Shangguan, 1996; Wassmann *et al.*, 2000). Increasing rice productivity can also enhance soil organic C stocks (Pan *et al.*, 2006). The mitigation potential of this measure is discussed further in section 4.9 below.

#### 4.2.6 Agro-forestry

Agro-forestry is the production of livestock or food crops on land that is also used to grow trees for timber, firewood, or other wood products. It includes shelter belts and riparian zones/buffer strips with woody species. The standing stock of carbon above ground is usually higher than the equivalent land use without trees, and planting trees may also increase soil carbon sequestration (Oelbermann *et al.*, 2004; Guo and Gifford, 2002; Mutuo *et al.*, 2005; Paul *et al.*, 2003). The effects on N<sub>2</sub>O and CH<sub>4</sub> emissions are not well known however (Albrecht and Kandji, 2003). Modelled estimates of the global mitigation potential of this measure are given in Table 6, and although not hugely significant at a global scale, it may play an important role in developing countries. Pandey (2002) provides a comprehensive review, while Makundi and Sathaye (2004) reviewed studies of C mitigation potential from seven developing countries with a focus on agro-forestry, and concluded that such practices could contribute up to 21 % of total

mitigation potential from forestry-related practices, and are particularly cost-effective. Albrecht and Kandji (2003) estimate that the C sequestration potential of agro-forestry systems is between 12 and 228 t ha<sup>-1</sup>, with a median value of 95 t C ha<sup>-1</sup>, meaning that such practices could store 1.1-2.2 Pg C over the next 50 years, the equivalent to removing between 4 and 8.1 Pg CO<sub>2</sub>-eq from the atmosphere (based on global area that is suitable for the practice of 585-1215 Mha).

#### 4.2.7 Land cover (use) change

One of the most effective methods of reducing emissions is often to allow or encourage the reversion of cropland to another type of land cover, typically one more similar to the native vegetation. This conversion can occur over the entire land area ('set-asides'), or in localized spots, such as grassed waterways, field margins or shelterbelts (Follett, 2001; Freibauer *et al.*, 2004; Lal, 2004b; Falloon *et al.*, 2004; Ogle *et al.*, 2003), and it often increases C storage. For example, converting arable cropland to grassland typically results in the accrual of soil C because soil disturbance is lower and C removal in harvested products is reduced. Compared to cultivated lands, grasslands may also emit less N<sub>2</sub>O due to lower N inputs, and also have higher rates of CH<sub>4</sub> oxidation, although recovery of this may be slow (Paustian *et al.*, 2004). Similarly, converting drained croplands back to wetlands can result in rapid accumulation of soil C (removal of atmospheric CO<sub>2</sub>). However, this conversion may stimulate CH<sub>4</sub> emissions, because water logging creates anaerobic conditions (Paustian *et al.*, 2004). Afforestation is another example of land use change to increase C storage. This type of land cover (or use) conversion comes at the expense of lost agricultural productivity, and is therefore usually an option only on surplus agricultural land or on croplands of marginal productivity. Table 6 gives modelled estimates of the global mitigation potential of this measure, and see Smith *et al.* (2007a).

As the global population is forecast to increase, rising food demand will drive increased agricultural productivity, and the more likely pressure on land use is towards more cropland rather than less. Just as removing land from agricultural production is a very effective measure for lowering GHG emissions, the reverse trend has the potential to significantly increase them. As discussed in section 2.1, CO<sub>2</sub> uptake and fluxes in agriculture are almost balanced currently (excluding indirect emissions from energy and fuel use), but land use change towards more cultivation may contribute a greater proportion of global total GHG emissions than direct emissions from the whole of the agricultural sector (6-17 % vs. 10-12 %). This is largely due to changes in the C budget when forest or

other natural/semi-natural habitats are converted to croplands.

Murty *et al.* (2002) reviewed studies into the effect of conversion of forest to either cropland or pasture and concluded that change to pasture caused on average no significant change in soil C stocks, although they reported changes ranging from -50 to +160 % depending on factors such as fertilizer applications and plant residue treatment. Conversion to cropland, reduced soil C in all but 11 observations out of the 75 surveyed. The average loss 10 or more years after conversion was 30 %, although the authors argue that failure to adjust for changes in soil bulk density resulted in around half of the studies over-estimating C losses, and discounting these gives a figure of 22 % loss of soil C. Searchinger *et al.* (2008) used a figure of 25 % soil C loss upon conversion from non-cropped to cropped land when considering the impact of biofuels expansion in the USA and admitted that this was conservative. In their critique of that study, ADAS (2008) state that a figure of 30-40 % may be more appropriate. This is based on studies such as Guo and Gifford (2002), whose meta-analysis of data from 74 publications gave average soil C losses of 42 % when native forest is converted to cropland and 59 % when pasture is converted to cropland, although they sound a note of caution regarding the exact magnitude of these losses given the wide range of methods used by the original investigations.

Nitrogen dynamics are more variable with conversion from forest to cultivated land, giving an average 15 % decrease in soil N stocks, and forest conversion to pasture having no significant effect on average. However, both increases and decreases in C:N ratios were reported (Murty *et al.*, 2002). When considering the GHG balance, it is important to remember that inputs of N in terms of fertilizer and animal manure will give rise to N<sub>2</sub>O emissions from these converted lands, which may be much higher than those from the forests or semi-natural grassland they replaced (Bouwman, 1990; Mosier *et al.*, 1991; Paustian *et al.*, 2004).

GHG emissions from converting temperate grassland to arable cropland have been estimated at 111-242 Mg CO<sub>2</sub>-eq ha<sup>-1</sup> over a 30 year period (IPCC, 2006; Searchinger *et al.*, 2008). A modelling study examining the impact of using formerly uncultivated land to compensate for short-falls in yields, calculated that compensating for a 30 % UK wheat yield reduction, could generate over 500 Tg CO<sub>2</sub>-eq over 20 years, depending on where in the world production was displaced to and whether it replaced forest or pasture (Carlton *et al.*, 2009). These figures strongly suggest that it is better to maximize yields on existing cropland vs. cultivating extra land.

### 4.3 Per-area mitigation potentials

As many practices can affect more than one GHG, it is important to consider the impacts of any potential mitigation options on all GHGs together (Robertson *et al.*, 2000; Smith *et al.*, 2001; Gregorich *et al.*, 2005). Table 6 is adapted from Smith *et al.* (2007b) and gives estimates of the potential impact of mitigation practices (except rice management) on all GHGs, on a per-area annual basis dependent on climatic conditions, based on the best available data. The estimates represent average change in soil C stocks (CO<sub>2</sub>) or emissions of N<sub>2</sub>O and CH<sub>4</sub> on a per-hectare basis, with positive values representing net CO<sub>2</sub> uptake increasing the soil C stock, or a reduction in emissions of N<sub>2</sub>O and CH<sub>4</sub>. These estimates were derived as follows:

**Estimates of soil C storage (CO<sub>2</sub> mitigation)** for all practices were derived from about 200 studies (see IPCC, 2006, Grassland and Cropland Chapters of Volume IV, Annexes 5A and 6A) using a linear mixed-effect modelling approach, which is a standard linear regression technique with the inclusion of random effects due to dependencies in data from the same country, site and time series (Ogle *et al.*, 2004, 2005; IPCC, 2006; Smith *et al.*, 2007c). The studies were conducted in regions throughout the world, but temperate studies were more prevalent leading to smaller uncertainties than for estimates for warm tropical climates. Estimates represent annual soil C change rate for a 20-year time horizon in the top 30 cm of the soil. Soils under agro-forestry were assumed to derive their mitigation potential mainly from cessation of soil disturbance, and given the same estimates as no-till.

**CH<sub>4</sub> emissions** for set-aside and land use changes (LUC) were based on modelling results from DAYCENT and DeNitrification-DeComposition (DNDC) (US-EPA, 2006b).

**N<sub>2</sub>O figures** for nutrient management were derived using the DAYCENT simulation model, and include both direct emissions from nitrification/denitrification at the site, as well as indirect N<sub>2</sub>O emissions associated with volatilization and leaching/runoff of N that is converted into N<sub>2</sub>O following atmospheric deposition or in waterways, respectively (US-EPA, 2006b; assuming a N reduction to 80% of current application). N<sub>2</sub>O figures for tillage and residue management were derived using DAYCENT (US-EPA, 2006b; figures for no-till).

As Table 6 illustrates, the effectiveness of mitigation options is variable across and within climate regions. Therefore, a practice that is very effective in reducing emissions at one site may be less effective or even counter-productive at another. As a result, it is not

possible to produce a universally applicable list of mitigation practices; instead, proposed practices need to be evaluated for individual agricultural systems according to the specific climatic conditions, historical land use and management, and other factors, such as social issues. The effectiveness of mitigation strategies also changes with time. Some practices, such as

those that promote soil C storage, have diminishing effectiveness after several decades; while others, such as increasing nutrient use efficiency, may reduce emissions indefinitely. For example, emissions from no-till agriculture show strong time dependency, in part because of changing influence of tillage on N<sub>2</sub>O emissions (Six *et al.*, 2004).

**Table 6.** Per-area annual mitigation potentials for each climate region for cropland mitigation options (Smith *et al.*, 2007b).

Climate zone	Practice	CO <sub>2</sub> (t CO <sub>2</sub> ha <sup>-1</sup> yr <sup>-1</sup> )		CH <sub>4</sub> (t CO <sub>2</sub> -eq ha <sup>-1</sup> yr <sup>-1</sup> )		N <sub>2</sub> O (t CO <sub>2</sub> -eq ha <sup>-1</sup> yr <sup>-1</sup> )		All GHG (t CO <sub>2</sub> -eq ha <sup>-1</sup> yr <sup>-1</sup> )	
		Mean	Range	Mean	Range	Mean	Range	Mean	Range
Cool-dry	Agronomy	0.29	0.07 - 0.51	0.00	0.00	0.10	0.00 - 0.20	0.39	0.07 - 0.71
	Nutrient management	0.26	-0.22 - 0.73	0.00	0.00	0.07	0.01 - 0.32	0.33	-0.21 - 1.05
	Tillage and residue management	0.15	-0.48 - 0.77	0.00	0.00	0.02	-0.04 - 0.09	0.17	-0.52 - 0.82
	Water management	1.14	-0.55 - 2.82	0.00	0.00	0.00	0.00	1.14	-0.55 - 2.82
	Set-aside and LUC	1.61	-0.07 - 3.30	0.02	0.00-0.02	2.30	0.00 - 4.60	3.93	-0.07 - 7.90
	Agro-forestry	0.15	-0.48 - 0.77	0.00	0.00	0.02	-0.04 - 0.09	0.17	-0.52 - 0.86
Cool-moist	Agronomy	0.88	0.51 - 1.25	0.00	0.00	0.10	0.00 - 0.20	0.98	0.51 - 1.45
	Nutrient management	0.55	0.01 - 1.10	0.00	0.00	0.07	0.01 - 0.32	0.62	0.02 - 1.42
	Tillage and residue management	0.51	0.00 - 1.03	0.00	0.00	0.02	-0.04 - 0.09	0.53	-0.04 - 1.12
	Water management	1.14	-0.55 - 2.82	0.00	0.00	0.00	0.00	1.14	-0.55 - 2.82
	Set-aside and LUC	3.04	1.17 - 4.91	0.02	0.00-0.02	2.30	0.00 - 4.60	5.36	1.17 - 9.51
	Agro-forestry	0.51	0.00 - 1.03	0.00	0.00	0.02	-0.04 - 0.09	0.53	-0.04 - 1.12
Warm-dry	Agronomy	0.29	0.07 - 0.51	0.00	0.00	0.10	0.00 - 0.20	0.39	0.07 - 0.71
	Nutrient management	0.26	-0.22 - 0.73	0.00	0.00	0.07	0.01 - 0.32	0.33	-0.21 - 1.05
	Tillage and residue management	0.33	-0.73 - 1.39	0.00	0.00	0.02	-0.04 - 0.09	0.35	-0.77 - 1.48
	Water management	1.14	-0.55 - 2.82	0.00	0.00	0.00	0.00	1.14	-0.55 - 2.82
	Set-aside and LUC	1.61	-0.07 - 3.30	0.02	0.00-0.02	2.30	0.00 - 4.60	3.93	-0.07 - 7.90
	Agro-forestry	0.33	-0.73 - 1.39	0.00	0.00	0.02	-0.04 - 0.09	0.35	-0.77 - 1.48
Warm-moist	Agronomy	0.88	0.51 - 1.25	0.00	0.00	0.10	0.00 - 0.20	0.98	0.51 - 1.45
	Nutrient management	0.55	0.01 - 1.10	0.00	0.00	0.07	0.01 - 0.32	0.62	0.02 - 1.42
	Tillage and residue management	0.70	-0.40 - 1.80	0.00	0.00	0.02	-0.04 - 0.09	0.72	-0.44 - 1.89
	Water management	1.14	-0.55 - 2.82	0.00	0.00	0.00	0.00	1.14	-0.55 - 2.82
	Set-aside and LUC	3.04	1.17 - 4.91	0.02	0.00-0.02	2.30	0.00 - 4.60	5.36	1.17 - 9.51
	Agro-forestry	0.70	-0.40 - 1.80	0.00	0.00	0.02	-0.04 - 0.09	0.72	-0.44 - 1.89

Note: Ranges indicate a 95 % confidence interval around the mean. Mean and uncertainty for change in soil C, N<sub>2</sub>O and CH<sub>4</sub> emissions are at the climate region scale, and are not intended to reflect finer scales such as individual farms.

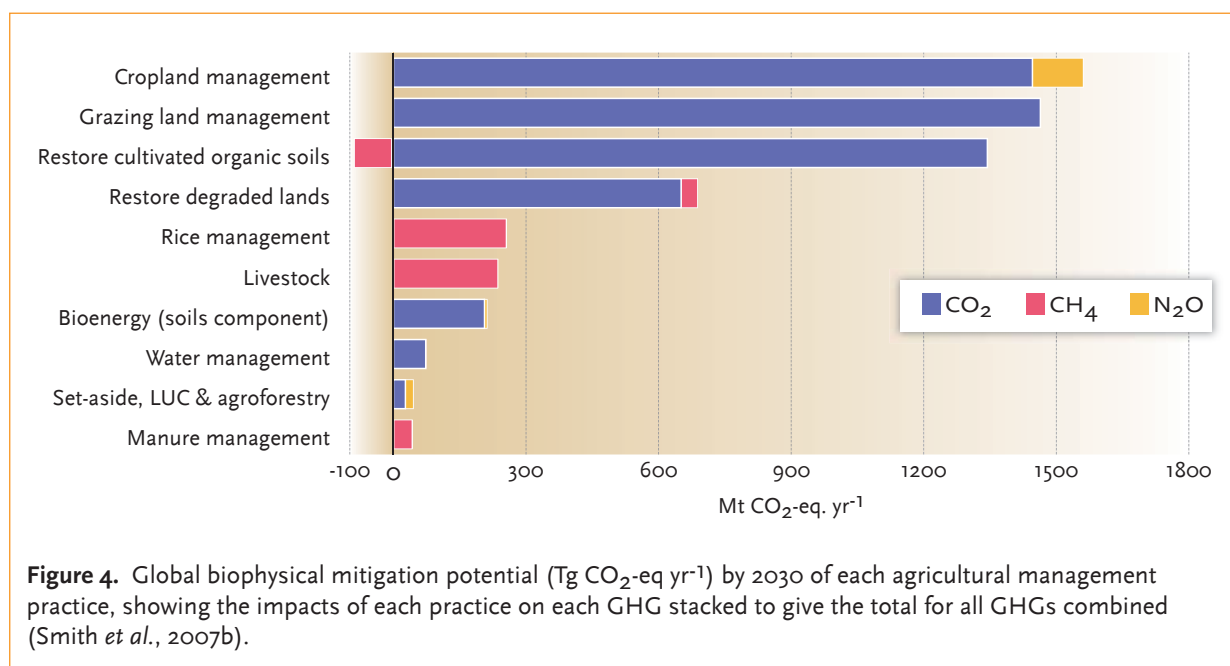


#### 4.4 Total global mitigation potentials

There have been numerous assessments of the potential for GHG mitigation in agriculture, mostly focusing on soil C sequestration. The IPCC Second Assessment Report (SAR; IPCC, 1996) estimated that 400-800 Tg C yr<sup>-1</sup> (equivalent to about 1400-2900 Tg CO<sub>2</sub>-eq yr<sup>-1</sup>) could be sequestered in global agricultural soils with a finite capacity saturating after 50 to 100 years. For other GHG, the report estimated that CH<sub>4</sub> emissions from agriculture could be reduced by 15 to 56 %, mainly through improved nutrition of ruminants and better management of paddy rice, and that improved management could reduce N<sub>2</sub>O emissions by 9-26 % (IPCC, 1996). The Third Assessment Report (TAR; IPCC, 2001) revised this to a total mitigation potential estimate of 350-750 Tg C yr<sup>-1</sup> (~1300-2750 Tg CO<sub>2</sub>-eq yr<sup>-1</sup>), with the range mainly caused by large uncertainties about CH<sub>4</sub>, N<sub>2</sub>O and soil-related emissions of CO<sub>2</sub>. The European Climate Change Programme (ECCP) Working Group on Sinks Relating to Agricultural Soils estimate that measures to promote C sequestration in agricultural soils in the EU-15 could mitigate 60-70 Tg CO<sub>2</sub> yr<sup>-1</sup>, equivalent to 1.5-1.7 % of anthropogenic CO<sub>2</sub> emissions for the region (ECCP, 2003). From other published estimates of technical potential, only Caldeira *et al.* (2004) and Smith *et al.* (2007b) have provided global estimates considering all GHGs together. Smith *et al.* (2007b) used per-area estimates of mitigation potential for each GHG (shown in Table 6 for cropland practices) and multiplied these by the area available for that practice, giving a total global

technical potential of ~5500-6000 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> by 2030, of which about 89 % is from soil C sequestration, about 9 % from mitigation of CH<sub>4</sub>, and about 2 % from mitigation of soil N<sub>2</sub>O emissions (see Figure 4, and note that the figure includes all mitigation practices associated with agriculture, so the majority of measures discussed in this report fall in the first column of cropland management measures). Caldeira *et al.* (2004) estimate the potential to ~4500 Tg CO<sub>2</sub>-eq yr<sup>-1</sup>. These global technical mitigation potentials do not consider economic or other barriers, and compare with estimated baseline emissions of around 8200 Tg CO<sub>2</sub>-eq by 2030 (Smith *et al.*, 2007a, using projected increases from US-EPA, 2006b).

When economic considerations are taken into account, estimates of mitigation potential can vary widely depending on both C price and the methodology used. Smith *et al.* (2007b) used marginal abatement cost curves to estimate global economic mitigation potentials for all GHGs for 2030 to 1500-1600, 2500-2700, and 4000-4300 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> at C prices of up to 20, 50 and 100 US\$ t CO<sub>2</sub>-eq<sup>-1</sup>, revising the mitigation potential of cropland management strategies from around 1450 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> shown in Figure 4 to around 750-850 Tg CO<sub>2</sub>-eq yr<sup>-1</sup>, across the same range of C prices. Other researchers have estimated global economic mitigation potentials for either N<sub>2</sub>O or CH<sub>4</sub> (US-EPA, 2006b) or for C sequestration (Manne and Richels, 2004), but not for all GHGs together. Most recently, agricultural abatement has been included in top-down global modelling of long-term climate stabilization scenario pathways, using a variety of alternative approaches



that have resulted in different decision modelling (Rose *et al.*, 2007). These top-down modelling studies have produced a wide range of mitigation potential estimates, which are shown in Table 7 (Note that the lack of an upper range for mitigation potentials at C prices above US\$ 50 is a product of the lack of estimates available for these prices, and that the N<sub>2</sub>O + CH<sub>4</sub> column is based on scenarios which considered both gases and not a total of the previous two columns).

## 4.5 N<sub>2</sub>O mitigation potential of improved nutrient management

It has been suggested that improved N management could increase N use efficiency (NUE) by 50 % (Smil, 2001; Erisman *et al.*, 2008), thus significantly decreasing future fertilizer requirements. The ECCP Working Group on Agriculture (WG7) consider that improving NUE through improved fertilizer management has the greatest potential for mitigating European agricultural N<sub>2</sub>O emissions (ECCP, 2001). With food, feed, fibre and bioenergy demand escalating, fertilizer use per unit of product should be reduced as far as agronomically and economically possible. The nutrient management options described here and shown in Table 6 reduce N fertilizer inputs by improving NUE rather than simply reducing application rates, which could lead to displacement of emissions if yields cannot be maintained (Carlton *et al.*, 2009). Improvements in nutrient management will not reduce fertilizer application rates where there are N deficiencies, or where the total soil N pool is being mined, threatening long-term soil health and sustainability.

### 4.5.1 Overview of factors affecting N use efficiency

The following sections, partly taken from a recent review by Snyder *et al.* (2009), examine factors that affect N use efficiency and measures that could be used to reduce N<sub>2</sub>O emissions from fertilization of crops, and, where possible, attempt to enumerate their mitigation potential.

#### 4.5.1.1 Nitrogen source

Snyder *et al.* (2009) provide a comprehensive review of the impact of different N sources on the resulting N<sub>2</sub>O emissions when applied to cropland, covering commercially-produced mineral fertilizers, biologically-fixed legume-derived N, animal manure, fertilizers stabilized with urease or nitrification inhibitors, and slow- or controlled-release fertilizers. Their findings are summarized below, with some additional information and discussion.

#### Commercially-produced fertilizers

There is disagreement in the literature as to whether different types of N produce different levels of N<sub>2</sub>O emissions. Some researchers have argued that urea produces the highest emissions, followed by ammonium-based fertilizers and then nitrate-based ones, although this may be affected by water conditions (Tenuta and Beauchamp, 2003; Velthoff *et al.*, 2003), while others have suggested there is insufficient evidence to support this claim (Eichner, 1990; Granli and Bøckman, 1994). Most recently, studies have highlighted how much other factors (e.g. tillage) can influence emissions from different types of N additions (Venterea *et al.*, 2005). It has been postulated that differences among fertilizer types almost disappear

**Table 7.** Global agricultural mitigation potentials for 2030 estimated using top-down models (adapted from Smith *et al.*, 2007a).

Carbon price (US\$ t CO <sub>2</sub> -eq <sup>-1</sup> )	Mitigation potential (Tg CO <sub>2</sub> -eq yr <sup>-1</sup> )		
	CH <sub>4</sub>	N <sub>2</sub> O	N <sub>2</sub> O + CH <sub>4</sub>
0-20	0-1116	89-402	267-1518
20-50	348-1750	116-1169	643-1866
50-100	388	217	604
>100	733	475	1208

Note: estimates for C prices below US\$ 50 are based on six scenarios, while those above are based on only one. Original sources: USCCSP, 2006; Rose *et al.*, 2007; Fawcett and Sands, 2006; Smith and Wigley, 2006; Fujino *et al.*, 2006; and Kemfert *et al.*, 2006.

once the rate of application, crop type, climate, SOC, soil pH, and length of experiment effects are taken into account (Stehfest and Bouwman, 2006), leading the authors to conclude that confounding variables prevent a valid comparison of N<sub>2</sub>O emission levels from different N sources (Snyder *et al.*, 2009). Table A1 in Appendix A includes reported N<sub>2</sub>O emission ranges from different fertilizer types, indicating that they can vary from below 0.1 % to over 40 % of the applied N.

#### **Biologically-fixed legume-derived N**

It has been suggested that using leguminous crops as part of a rotation could be a mitigation measure (see section 4.2.1 above) as it reduces the need for fertilizer N additions. Although this fixed N is generally less available for nitrification, subsequent denitrification, and therefore N<sub>2</sub>O emission while the crop is actively growing, once it is harvested or dies, N-rich residues are left above and below ground to decompose and emit N<sub>2</sub>O. Several researchers have observed that background N<sub>2</sub>O emissions are increased by N-fixing plants (Rochette *et al.*, 2004; Dick *et al.*, 2006; Mosier *et al.*, 2006) and, therefore, the mitigation potential may be most dependent on the level of emissions from the fertilizer applied to the comparison crop (see for example, Parkin and Kasper, 2006), which can be highly variable. Productivity of a complete rotation might be reduced if the legumes take up space that would otherwise be available for the main crop (i.e. unless they are used as an intercrop or are undersown). In such cases, GHG emissions per unit of crop production, rather than per unit area, may not be reduced. However, if rotation productivity is maintained, GHG emissions per-unit-area and per-unit-product could be reduced.

#### **Animal manure**

Several researchers have reported that manure additions lead to greater N<sub>2</sub>O emissions than mineral fertilizers. For example Lowrance *et al.* (1998) found that high rates of liquid manure application produced denitrification rates 10-100 times higher than those from inorganic N fertilizer applications, and Velthof *et al.* (2003) also found that manure gave higher emissions, which were dependent on type and quality, as well as how they were handled. Snyder *et al.* (2009) conclude that there are many factors preventing more use of manure as a substitute for mineral fertilizer, including synchronization of nutrient release with crop demand, spatial coordination and planning at the regional level, and technological advances (Oenema *et al.*, 2001; Oenema and Pietrzak, 2002). Furthermore, transportation and labour costs associated with handling can make manure nutrients less economically attractive than fertilizers. However,

as discussed in section 2.3.2, not all researchers consider that animal manure applications lead to higher N<sub>2</sub>O emissions than mineral fertilizers. In fact, Ball *et al.* (2004) argue that organic manures do not cause the spikes in emissions that occur with mineral fertilizers if there is heavy rainfall around the time of application, and that they can therefore mitigate N<sub>2</sub>O emissions by up to around 90 % over a growing season. There is also the issue of alternative disposal methods and what emissions these can lead to, as at least when manure is applied to agricultural land it can offset fertilizer requirements. The ECCP Working Group on Sinks Relating to Agricultural Soils (ECCP, 2003) consider that promoting organic inputs to arable land would be the most effective measure for increasing C sequestration in European soils because it can be applied to such a large area (see section 4.6), and Vergé *et al.* (2007) also consider the practice beneficial, particularly in areas such as Africa, where tuber crops that return little organic residue to the soil cover a large percentage of agricultural land and have a negative impact on soil organic matter and fertility. Therefore, the best strategies are likely to be integrated ones that make the best use of both organic and mineral N resources.

#### **Slow- and controlled-release fertilizers**

Slow- and controlled-release fertilizers work through a variety of mechanisms, e.g. controlled water solubility of the material (by semi-permeable coatings, occlusion, inherent water insolubility of polymers, natural nitrogenous organics, protein materials, or other chemical forms), or by slow hydrolysis of water-soluble low molecular weight compounds (AAPFCO, 1995 cited in Weiske, 2006). By affecting the timing of N release from fertilizer, these compounds have the potential to reduce leaching and volatile losses of N, as well as N<sub>2</sub>O emissions (Shaviv, 2000; Halvorson *et al.*, 2009), and therefore may improve NUE and provide greater stability in fertilizer N performance. Several studies have demonstrated that controlled-release fertilizers reduce N<sub>2</sub>O emissions from fertilized soils in comparison with standard mineral fertilizers (Maggiato *et al.*, 2000; Motavalli *et al.*, 2008; and see Table 8), although some have suggested that emissions are only reduced around the time of application, when standard N applications give a spike of emissions, and continue over a longer period to give higher total N<sub>2</sub>O emissions (Delgado and Mosier, 1996). This is supported by field studies indicating that N<sub>2</sub>O emission spikes can occur more than a month after urea application if a polymer coating is used (Halvorson *et al.*, 2009). However, it should be noted that there is also still a degree of uncertainty about these fertilizers in terms of yield, quality and NUE. It has been argued that certain slow- or

controlled-release fertilizers are less able to meet crop nutrient demand when it is most needed compared with strategies based on multiple applications tailored to crop demand.

#### **Urease and nitrification inhibitors**

Soluble fertilizers formulated with inhibitors reduce or block the conversion of N species by affecting specific types of microbes involved in the N cycle. Urease inhibitors delay the enzymatic hydrolysis of urea (Watson, 2005; Rawluk *et al.*, 2001) while nitrification inhibitors block or control conversion of ammonium into nitrates. This helps to keep N in the form of ammonium for longer, thus encouraging uptake by crops and helping to prevent N<sub>2</sub>O emissions

from either nitrification or denitrification. Weiske (2006) summarized studies documenting how the use of different nitrification inhibitors can reduce N<sub>2</sub>O emissions from mineral fertilizers (see Table 8). However, it is important to recognize that a number of these studies were conducted over relatively short time periods and, therefore, may over-estimate seasonal or annual N<sub>2</sub>O emission reductions, as short-term point estimates are considered inadequate for determining these by some researchers, who recommend continuous, daily measurements over long time periods (Thornton and Valente, 1996).

It is obvious from the above that relationships between N<sub>2</sub>O emissions and N additions are complex and variable, meaning it is not possible to make firm

**Table 8.** Reported effectiveness of nitrification inhibitors and urea coatings (partially adapted from Weiske, 2006).

Type of inhibitor or coating	Fertilizer type	Crop	Length of monitoring	% N <sub>2</sub> O emission reduction	Reference
Nitrapyrin	Ammonium sulphate	Lab study, soil only	30 days	93	Bremner and Blackmer, 1978
Nitrapyrin	Urea	Lab study, soil only	30 days	96	Bremner and Blackmer, 1978
Nitrapyrin	Urea	Corn	100 days	40-65	Bronson <i>et al.</i> , 1992
Calcium carbide	Urea	Corn	100 days	33-82	Bronson <i>et al.</i> , 1992
DCD	Liquid manure	Pasture grass	14 days	50-88	De Klein and van Logtestijn, 1994
DCD	Ammonium sulphate	Pasture grass	64 days	40-92	Skiba <i>et al.</i> , 1993
DCD	Urea	Spring barley	90 days	82-95	Delgado and Mosier, 1996
DCD	Urea	Spring barley	1 growing season	81	Shoji <i>et al.</i> , 2001
DCD	Urea	Wheat	95 days	49	Majumdar <i>et al.</i> , 2002
DCD	Urea	Spring barley	56 days	40	McTaggart <i>et al.</i> , 1997
POCU	Urea	Spring barley	90 days	35-71	Delgado and Mosier, 1996
POCU	Urea	Spring barley	1 growing season	35	Shoji <i>et al.</i> , 2001
DCS	Ammonium sulphate	Pasture grass	64 days	62	Skiba <i>et al.</i> , 1993
DMPP	Ammonium sulphate nitrate	Spring barley, corn and winter wheat	3 years (spring and summer only)	51	Weiske <i>et al.</i> , 2006
'Neem' coating	Urea	Wheat	95 days	9	Majumdar <i>et al.</i> , 2002
'Nimin' coating	Urea	Wheat	95 days	63	Majumdar <i>et al.</i> , 2002
Thiosulphate	Urea	Wheat	95 days	35	Majumdar <i>et al.</i> , 2002
Polymer coating	Urea	Corn (no-till)	159 days	55	Halvorson <i>et al.</i> , 2009

DCD = dicyandiamide

POCU = polyolefin coated urea

DCS = N (2,5 dichlorophenyl) succinic acid monoamide

DMPP = 3,4-dimethylpyrazole phosphate



recommendations on fertilizer types. For example, while Halvorson *et al.* (2009) report polymer coating reduced N<sub>2</sub>O emissions from urea by 55 % over the course of a growing season for continuous no-till corn, the same experiment carried out on conventionally tilled continuous corn gave no significant difference in cumulative N<sub>2</sub>O emissions between coated and uncoated urea. However, nitrification inhibitors and, maybe to a lesser extent, controlled- or slow-release fertilizers, do appear to have potential for mitigating future emissions, although further research is needed to quantify this potential over longer periods of time.

#### 4.5.1.2 Rates of application

Use of appropriate N application rates limit the build up of nitrates in the soil, which can accumulate when more N is applied than needed by the crop or when applications are poorly timed to optimize crop uptake (Legg and Meisinger, 1982). N application rates cannot just be cut below economic optimums without risking long-term decline in soil productivity (Jaynes and Karlen, 2005), but managing N fertilizer rates to an appropriate amount is challenging because each agro-ecosystem and specific growing season will differ as to what is appropriate. As Snyder *et al.* (2009) point out in their review, wide ranges in yield response to a given N rate often occur when attempting to calibrate soil nitrate tests or mineralizable N tests (Follett, 2001; Dahnke and Johnson, 1990; Stanford, 1982), with a scatter of data points instead of a distinct response curve because of the large spatial and temporal variability (Meisinger, 1984).

Many studies have suggested that when an agronomic N threshold level is exceeded, N<sub>2</sub>O emissions can increase dramatically (see e.g. Kachanoski *et al.*, 2003; Grant *et al.*, 2006; Malhi *et al.*, 2006). Grant *et al.* (2006) argue that this increase in proportional loss occurs when fertilizer applications exceed ecosystem N uptake capacity over time, which can be estimated from pre-planting measurements of residual N, and from annual estimates of gains or losses in soil organic N and of removals in harvest N; much the same as estimates of N requirements are made. However, there is disagreement in the literature as to whether the relationship between N inputs and N<sub>2</sub>O emissions remains constant at higher application levels; the IPCC default methodology assumes a linear response, which agrees with irrigated cropping system research in Colorado (Mosier *et al.*, 2006; Halvorson *et al.*, 2008), while some researchers have argued that the rate of increase in emissions tails off at higher application rates (Thornton and Valente, 1996). However, most evidence is in favour of a non-linear response whereby emissions are relatively constant across a broad range of application rates, and then tend to increase at the highest application levels (Kachanoski *et al.*, 2003; McSwiney and Robertson,

2005; Grant *et al.*, 2006), a conclusion reached by Bouwman *et al.* (2002) after reviewing data from across multiple sites, years, sources and cropping systems.

In practical terms, a farmer can minimize the potential for N<sub>2</sub>O emissions by following a nutrient management plan (NMP) that includes determining N inputs from residual nitrate levels and normal N mineralization potential from SOM in the field soils, and then filling in the deficit between the sum of these two N inputs and reasonable crop yields with an appropriate amount of animal manure and timely and well-placed N fertilizer (Snyder *et al.*, 2009). Split applications of N are already standard practice in Europe and allow for fine-tuning of rates by in-season plant analysis.

#### 4.5.1.3 Application placement

Some researchers have suggested that N<sub>2</sub>O emissions are greater when some N fertilizer applications are placed at greater depth. Breitenbeck and Bremner (1986) reported that emissions from a soil fertilized with anhydrous ammonia at a rate of 112 kg N ha<sup>-1</sup> were 107 % and 21 % greater when injected at a depth of 30 cm than at depths of 10 cm and 20 cm, respectively, although this effect was reduced when this fertilizer was applied at a rate of 225 kg N ha<sup>-1</sup>, while Drury *et al.* (2006) found emissions were up to around 60 % higher when ammonium nitrate used to fertilize corn was placed at 10 cm depth rather than 2 cm, depending on tillage treatment (although results varied more between years than treatments). Others, however, have found no effect of depth (Venterea and Stanenas, 2008). Snyder *et al.* (2009) argue that when urea-containing N sources are applied on the soil surface and not incorporated, a substantial proportion is lost via volatilization of ammonia (NH<sub>3</sub>), especially with manure or urea in humid environments, which reduces N levels available for plant uptake without reducing N<sub>2</sub>O emissions. Therefore, best practice measures which reduce NH<sub>3</sub> volatilization also reduce N<sub>2</sub>O emission in the same proportion as the amount of N conserved. Hultgreen and Leduc (2003) report that when urea was applied in a band below and to the side of the seed row, emissions were reduced in comparison to broadcast surface application in two years of a three-year study at two sites in Saskatchewan.

#### 4.5.1.4 Application timing

N applications carefully timed to maximize crop uptake have been shown to allow lower application rates of fertilizer without any yield loss and to reduce N<sub>2</sub>O emissions. Matson *et al.* (1998) reported that changing the application of 250 kg N ha<sup>-1</sup>, split 75 % a month before planting and 25 % a month after, to 180 kg N ha<sup>-1</sup>, split 33 % at planting and 67 % six weeks later, produced the same yield and 50 % less

$N_2O$  emissions, although it is difficult to assess how much of this reduction is due to timing and how much to the reduction in N rates, which may have been excessive initially. Hultgreen and Leduc (2003) showed that  $N_2O$  emissions are lower from fertilizer applied in spring rather than autumn. Snyder *et al.* (2009) pointed out that, in a study by Zebarth *et al.* (2008), available nitrate levels were a poor indicator of variations in  $N_2O$  emissions, as reducing application rates and changing to side-dress application reduced nitrate levels but not  $N_2O$  emissions, when fertilizer applications had previously been in excess of crop demand. This suggests that the strategy may not work for intensively managed crops, or that there may be a lag period, and suggests further research is needed in this area. Wagner-Riddle *et al.* (2007) suggest that timing N applications to match crop uptake combines well with no-till management to reduce  $N_2O$  emissions.

#### 4.5.1.5 Balanced fertilization with other required nutrients

Research has shown that balancing N applications with other required nutrients, such as phosphorus (P), potassium (K) and sulphur (S), improves NUE (Schlegel *et al.*, 1996; Johnson *et al.*, 1997; Gordon, 2005), as nitrates may accumulate in the soil if factors other than N availability are limiting crop growth, and this form of N is mobile and prone to environmental loss. Snyder *et al.* (2009) argue that balanced fertilization and soil fertility are major factors by which farmers can control NUE, and a recent review showed that the average fertilizer N recovery ((fertilized crop N uptake - unfertilized crop N uptake)/N applied) across studies in China, India and North America was 54 % for balanced fertilization treatments vs. 21 % for conventional or control treatments (Fixen *et al.*, 2005). As well as being a key component of fertilizer best management practices (BMPs), balanced fertilization is crucial for sustainable, high yielding, good quality crop production (Cissé, 2007).

#### 4.5.1.6 Fertilizer best management practices

Drawing on all the information summarized above, Snyder *et al.* (2009) produce an extensive list of BMPs, many of which were identified by the International Fertilizer Industry Association taskforce report on reactive N (IFA, 2007) and highlighted by Snyder (2008), based on the principles of good fertilizer stewardship –

the right source, at the right rate, at the right time, and at the right place (Roberts, 2007). These include recommendations for general practices, such as using a nutrient management plan and timing applications to maximize crop uptake, and also recommendations to do with application equipment and crop rotation management (see Snyder *et al.*, 2009, section 4).

### 4.5.2 $N_2O$ Mitigation potential case studies

#### Improving nutrient management in China

According to IFADATA (2009), total mineral N fertilizer consumption in China was 31.8 Tg N in 2006, which is more than the total for all developed countries, and has been increasing, though not consistently, for the past 20 years (Figure 5). Although China is a vast country with an arable and permanent crop area of nearly 153 Mha for 2007 (FAOSTAT, 2009), it has such a high population that the arable area per capita is well below the global average, leading to the development of highly intensive agriculture in order to feed its populace (Zhu and Chen, 2002). This huge consumption is also due to per area application rates that are much higher than the world average (Yang, 2006), up to 1900 kg N ha<sup>-1</sup> yr<sup>-1</sup> according to Zhang *et al.* (1996). This is partially made possible by the cost of fertilizers having been very low, achieved by subsidies for production and distribution, and also the fact that the high number of small-scale farmers make it difficult to monitor fertilizer use and enforce more sustainable practices (Zeng, 2003). The result is that NUE, as defined by crop output produced per unit of N applied (partial factor productivity), for cereal crops is considered to have declined since the 1980s and is estimated to be around 23 kg cereals kg<sup>-1</sup> N, in comparison with a global average of around 33 kg cereals kg<sup>-1</sup> N (IFA, 2007). However, this was calculated assuming two-thirds of fertilizer is applied

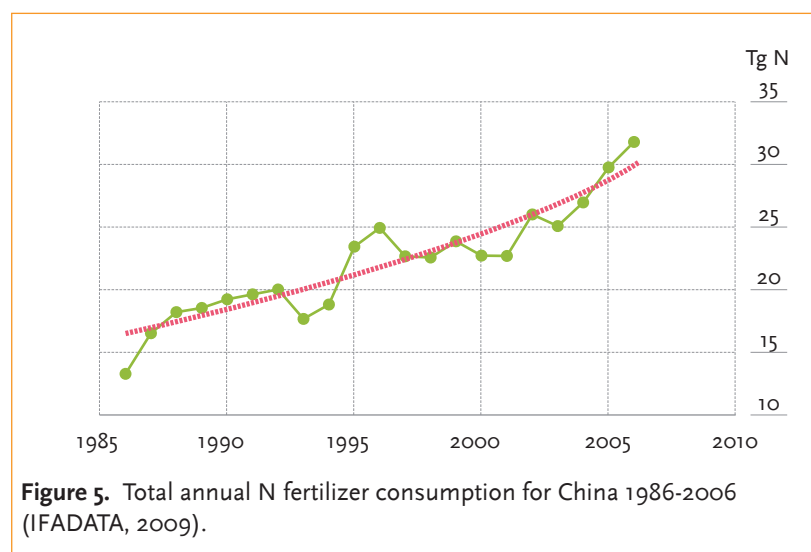


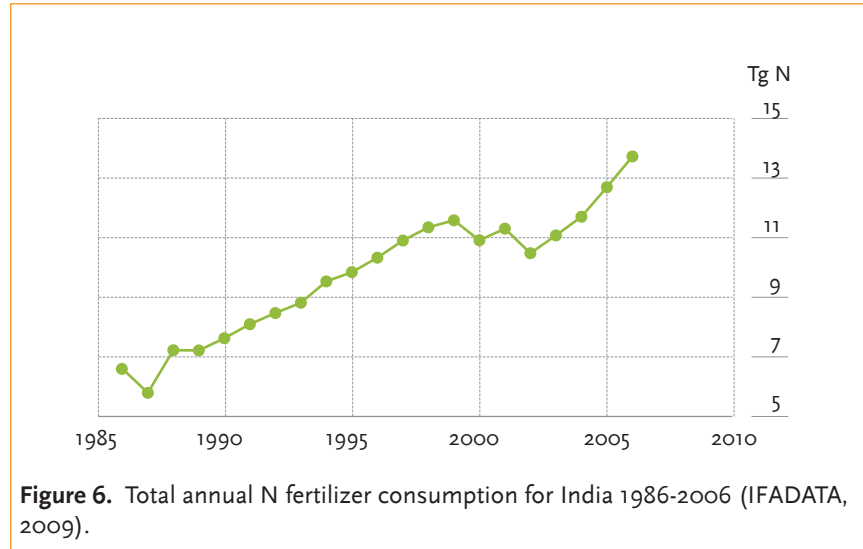
Figure 5. Total annual N fertilizer consumption for China 1986-2006 (IFADATA, 2009).

to cereals, which is an overestimate as fruit and vegetable production in China has shown a rapid increase (IFA, 2007), and the latest figures estimate that just under 50 % of fertilizer N used in China 2006 was applied to cereals (Heffer, 2009).

Calculations for individual crops with known fertilization rates indicate that NUE, as measured by partial factor productivity, is 12 kg wheat kg N<sup>-1</sup> and 17 kg maize kg N<sup>-1</sup> for China, in comparison with 31 kg wheat kg N<sup>-1</sup> for France and 27 kg maize kg N<sup>-1</sup> for the USA (Zhang *et al.*, 2009).

Regardless of the exact NUE, it is clear that the significant increases in fertilizer N usage in China have not resulted in proportional increases in food production. Zhu and Chen (2002) reported that regression analysis indicates the apparent efficiency of manufactured fertilizer N has been dropping rapidly over the past 50 years. Chen *et al.* (2004) report applications as high as five times the crop requirement and this overuse of mineral N fertilizers means there is vast scope for reductions in related N<sub>2</sub>O emissions without a loss of yield.

Zhu and Chen (2002) ascribe the low efficiency of fertilizer applications in China to a combination of overly high application rates, improper application methods, applications too early in the season when crop N uptake capacity is small, unbalanced fertilization (particularly lack of K additions), and use of inappropriate N forms that are more prone to loss. The study goes on to discuss a range of BMP options, such as those described above, in the Chinese context. Furthermore, the study cited field data that indicates that deep placement of urea can reduce N loss by at least 12 % (Zhang *et al.*, 1992; Zhu, 1998; Cai *et al.*, 2002), deep placement and split applications can reduce application rates by 7-24 % without yield loss (Zhu, 1997), and nitrification inhibitors can reduce denitrification losses of applied N by 21-27 % (Chen *et al.*, 1994). More recently, a large scale project involving the development and dissemination of fertilizer BMPs has shown that, compared with farmers' traditional practices, BMPs can, on average, reduce N inputs by 20-40 %, increase yields by 2-12 %, increase N recovery rates by 10-15 %, and reduce N losses by 10-50 %, over nine cropping systems across China (Zhang *et al.*, 2007). Researchers also report that better use of organic nutrient sources such as manure, could reduce the requirement for mineral N fertilizers by about 9 Tg (Zhang *et al.*, 2009).



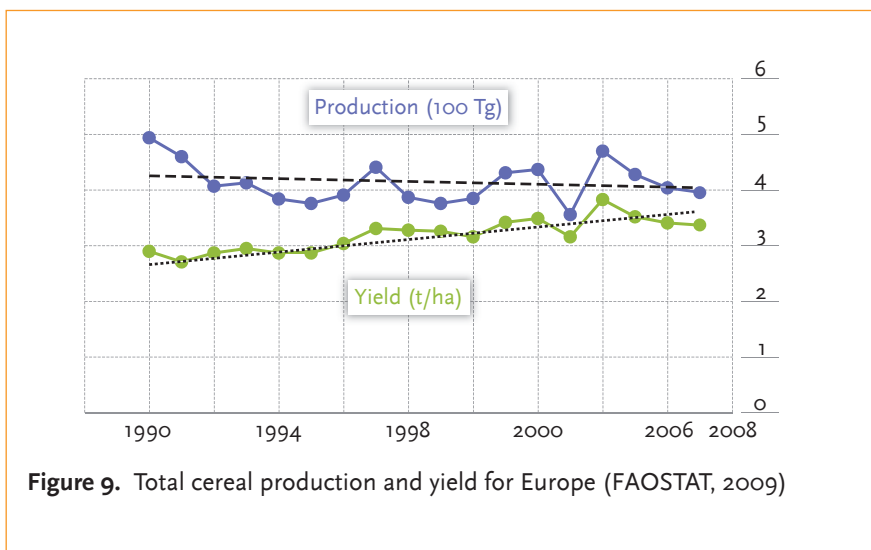
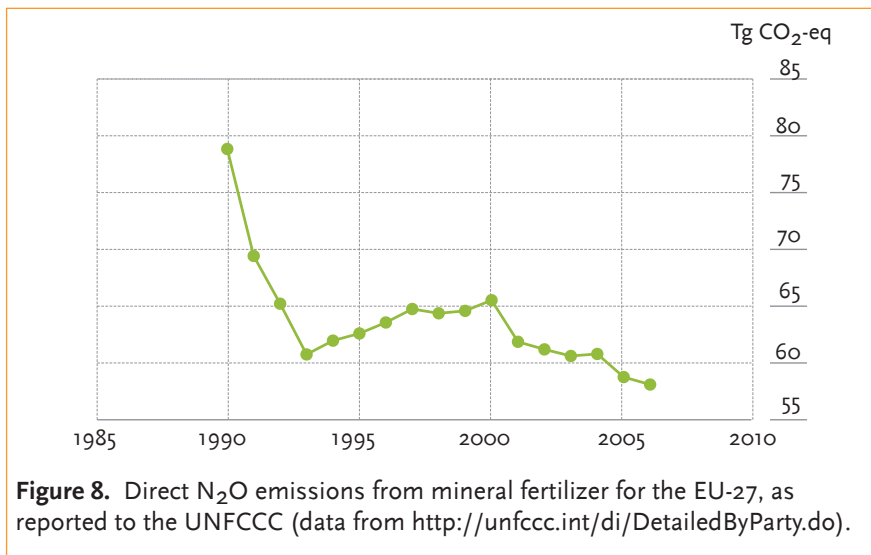
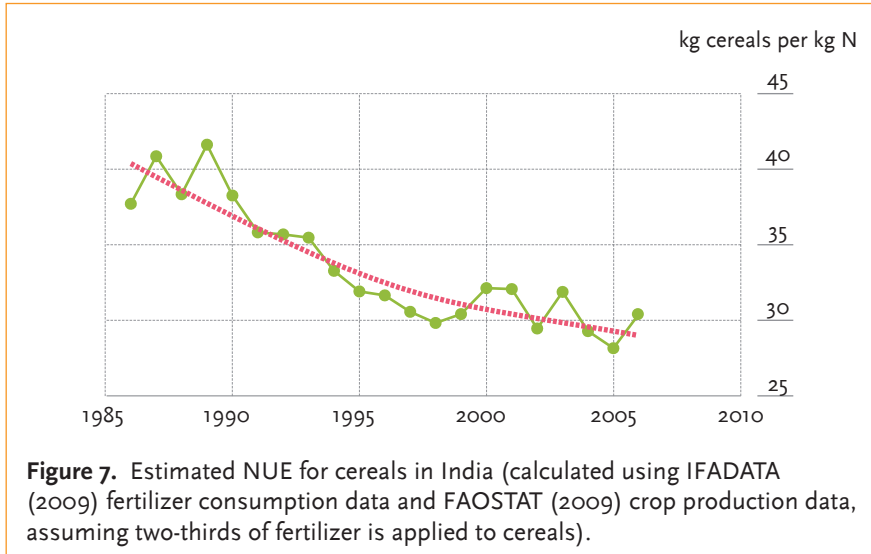
Any measure that reduces fertilizer manufacture in China will also have a large effect in reducing global GHG emissions from this activity. This is because 97 % of ammonia produced using coal is manufactured in China and these production plants are less energy-efficient than those that use gas and predominate in the rest of the world.

#### **Improving nutrient management in India**

According to IFADATA (2009), India is the second largest consumer of N fertilizer after China, with consumption totalling 13.8 Tg N in 2006. N fertilizer use has been increasing steadily for the past 20 years, with the exception of a short period around the turn of the 21st Century (Figure 6).

Over the same period, NUE for cereals, measured by partial factor productivity, has declined from around 40 kg cereals produced per kg of N fertilizer applied in the late 1980s, to only 26 kg cereals kg<sup>-1</sup> N in 2006. Although it seemed to be stabilizing in the late 1990s (IFA, 2007), Figure 7 shows a declining trend in more recent years. These figures were calculated assuming two-thirds of N fertilizer is applied to cereals, whereas the latest data (Heffer, 2009) estimate that the figure was only 58 % for 2006/07. This revises the NUE estimate to 30 kg cereals kg<sup>-1</sup> N for 2006, which is still below the global average.

This trend of declining NUE illustrates the combined effect of very non-uniform patterns of regional fertilizer use with some highly intensive cropping areas, for example some regions of north-western India with application rates as high as 300 kg N ha<sup>-1</sup> yr<sup>-1</sup>, and other areas where chronic N-deficiency is still a major problem (Galloway *et al.*, 2008). Therefore, improved nutrient management would actually involve increasing fertilizer applications in some areas, meaning that national fertilizer consumption may not be reduced. Indeed, given the need to feed



a burgeoning population, Singh and Singh (2008) consider that there is little opportunity to reduce total fertilizer use in India, and they argue that maximum emphasis should be given to improving NUE, particularly for wheat and rice cropping systems. The study goes on to discuss how this might be achieved, focusing on improving the synchrony between crop N demand and supply via methods such as precision farming, split applications and use of controlled-release fertilizers, and it concludes that there is substantial scope for improvement. Using mitigation scenarios, Garg *et al.* (2004) calculated that India could reduce N<sub>2</sub>O emissions in 2030 by up to 25 % compared to a business-as-usual scenario. This would be partially achieved by ensuring nitrate fertilizers are applied to aerobic systems while ammonium fertilizers are used for wetland crops, and by using nitrification inhibitors.

#### **Nutrient management in Europe and the USA**

Europe has already reduced fertilizer consumption. IFA data indicates that total N fertilizer consumption for Western and Central Europe has dropped by 24 % over the past 20 years: from 14.24 Tg N yr<sup>-1</sup> in 1985 to 10.88 Tg N yr<sup>-1</sup> in 2006 (IFADATA, 2009). This is largely due to the collapse of the former Soviet Union. Although the areas do not match up precisely and the data only go back to 1990, national reports to the UNFCCC for the EU-27 show a corresponding 27 % decline in fertilizer-induced N<sub>2</sub>O emissions between 1990 and 2006, albeit mostly achieved in the first four years (Figure 8).

As shown in Figure 9, over the same time period, total cereal production in Europe decreased slightly on average but yields per hectare, despite being variable, show an increasing trend overall. Assuming cereal fertilization rates per hectare have not increased, this suggests that reductions in total fertilizer use and associated emissions have, at least in part, been due to improved nutrient management. Further savings are still achievable. The ECCP (2001) estimated that N<sub>2</sub>O emissions from agricultural soils in the EU-15 could be mitigated by up to 50 Tg CO<sub>2</sub>-eq yr<sup>-1</sup>, with up to 10 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> achievable by 2012 through measures with little or no cost. Studies from the Netherlands have suggested that fertilizer rates could be reduced by 25 % in comparison with those used in 2000 without reducing crop yields, and that improved timing and split application could reduce fertilizer requirements on arable land by 10 % (Velthoff *et al.*, 2000a,b), while Godwin *et al.* (2003) argue that the most advanced precision farming systems can reduce fertilizer usage by 30 %. In the context of 2006 levels, a reduction of 10-30 % in total fertilizer N consumption across the whole of the EU-27 could reduce direct fertilizer-induced N<sub>2</sub>O emissions by 5.8-17.5 Tg CO<sub>2</sub>-eq., assuming a proportional decrease in N<sub>2</sub>O emissions.

In the USA, NUE for maize, for example, has shown a trend of striking improvement since the 1970s (Fixen and West, 2002), and there is still much scope for improvement. Recent work by Halvorson and others in Colorado (2009) has shown 20-50 % reductions in N<sub>2</sub>O emissions are possible with enhanced-efficiency fertilizer N use. If these technologies alone were used, reductions in N<sub>2</sub>O emissions associated with fertilizer N BMPs in the USA alone could equate to 9.6 to 24.1 Tg CO<sub>2</sub>-eq yr<sup>-1</sup>, as the latest US inventory estimated N<sub>2</sub>O emissions from fertilizer N applied to croplands and grasslands as 48.2 Tg CO<sub>2</sub>-eq for 2007 (US-EPA, 2009).

#### **4.6 Carbon sequestration potential of improved nutrient management**

Improved nutrient management can involve increasing nutrient applications to agricultural systems that are currently nutrient poor. Doing this in the form of mineral N fertilizers will increase productivity in N limited systems and, therefore, may increase C storage or sequestration (Desjardins *et al.*, 2005), while manure applications add additional organic matter to the soil directly, as well as acting to increase productivity. The ECCP Working Group on Sinks Related to Agricultural Soils (ECCP, 2003) consider that promoting organic inputs to arable land would be the most effective measure for increasing C sequestration in European soils, with a per-area mitigation potential of 1-3 t CO<sub>2</sub> ha<sup>-1</sup> yr<sup>-1</sup>, but a total potential (for the EU-15 over the first Kyoto commitment period, which is up to 2012) of 20 Tg CO<sub>2</sub> yr<sup>-1</sup> because it can be applied to such a large area. Similarly, a review (Desjardins *et al.*, 2005) cites data indicating a mitigation potential of 0.7-1.8 t CO<sub>2</sub> ha<sup>-1</sup> yr<sup>-1</sup> for manure applications, although a lower range for mineral fertilizer (see Table 9). Other researchers have suggested that while manure additions help to conserve soil C, mineral fertilizer applications have little effect on C sequestration in cropland systems rather than managed grasslands (Bertora *et al.*, 2009). However, in the Broadbalk winter wheat experiment at Rothamsted, Johnston *et al.* (2009) found that when N rates were increased to 240 and 288 kg N ha<sup>-1</sup> in 1985, soil C increased by about 16 %, to 1.22 and 1.29 % C, respectively, having been in equilibrium at around 0.93 and 1.12 % C under the previously lower rates of fertilizer N application. In the Midwestern US, Jagadamma *et al.* (2007) also found that soil organic carbon (SOC) sequestration rates increased at 0 to 30 cm and 0 to 90 cm with increasing N rates, compared to no N. They noted that C emissions associated with the production, transport and application of fertilizers must also be considered and concluded that proper fertilization, coupled with adoption of reduced tillage methods and integrated nutrient management,



**Table 9.** Reported C sequestration in response to nutrient additions (adapted from Desjardins *et al.*, 2005).

Addition type, region	C sequestration (t CO <sub>2</sub> ha <sup>-1</sup> yr <sup>-1</sup> )	Reference
Mineral fertilizer, USA	0.18-0.55	Lal <i>et al.</i> , 1998
Mineral fertilizer, USA	0.51-0.66	Halvorson <i>et al.</i> , 1999
Mineral fertilizer, USA	0.59	Lee and Dodson, 1996
Mineral fertilizer, Canada	0.15	Smith <i>et al.</i> , 2001
Mineral fertilizer, various (42 data points)	1.10	Conant <i>et al.</i> , 2001
Manure, USA	0.73-1.83	Follett, 2001b

may result in net SOC sequestration. Despite these findings, a global meta-analysis of numerous long-term fertilization experiments showed a minimal long-term effect of mineral N fertilization on soil C storage in croplands (Glendining and Powlson, 1995).

As already discussed, fertilizer and manure additions tend to increase N<sub>2</sub>O emissions, which can offset the mitigating effect of increased C storage. Desjardins *et al.* (2005) argue that very few studies to date have examined both C sequestration and other GHG emissions in the same field trials and, therefore, it is difficult to predict the effect on the overall GHG balance, although modelled data presented in the study suggests an 150 % increase in N fertilization may not significantly affect cumulative emissions from Canadian wheat rotations. Assuming a 1 % EF, mineral fertilizer applications of 118 kg N ha<sup>-1</sup> yr<sup>-1</sup> would be sufficient to offset a sequestration rate of 0.55 t CO<sub>2</sub> ha<sup>-1</sup> yr<sup>-1</sup> in terms of GWP, which is not a particularly high amendment level on a global scale, being, for example, within the range of additions for Scottish agricultural soils calculated by Flynn *et al.* (2005). The high degree of variation in N<sub>2</sub>O emissions from different fertilizers and management practices may be the main factor influencing the overall value of this mitigation practice, meaning it must be assessed on an individual basis for specific circumstances and cannot be easily generalized. As a final point, there is a saturation point beyond which additional C inputs will not result in increased long-term storage, and C sequestration can easily be reversed by a change in management (Desjardins *et al.*, 2005) while N<sub>2</sub>O emission savings are permanent.

#### 4.7 Mitigation potential of tillage and residue management

As discussed in section 4.2.3 above, reducing disturbance of the soil limits aeration and can therefore slow decomposition processes and reduce losses of soil C and GHG emissions. Zero-tillage (no-till) systems represent an extreme form of cropland management, in which any form of mechanical soil

disturbance is abandoned except for shallow opening of the soil for seeding, e.g. continuous mulch-seed or direct-drill. Reduced or conservation tillage covers a range of management options, where tillage may be reduced in frequency and/or depth to cause less soil disturbance than conventional ploughing. It includes ridge tillage (in which ridges are made in the field), shallow ploughing and rotovation or scarification of the soil surface. The timing of tillage can also be taken into account as part of this measure because autumn ploughing leaves fields bare over the winter, increasing the impact in terms of soil erosion and, in colder climates, N<sub>2</sub>O emissions from freeze-thaw cycles (Kaiser *et al.*, 1998; van Bochove *et al.*, 2000). Therefore, spring ploughing may reduce soil C losses and GHG emissions from tillage. In comparison to these measures, which reduce C losses, incorporating crop residues into soils after harvest works by adding C to the system.

##### 4.7.1 Zero tillage

Many studies have investigated the effectiveness of zero tillage for improving C storage in arable soils, reporting SOC increases of 7 % yr<sup>-1</sup> (global data set - West and Post, 2002), and 10-13 % for dry temperate regions and 16-18 % for moist temperate regions (Ogle *et al.*, 2005). Desjardins *et al.* (2005) provide a summary of reported C sequestration values from around the world (Table 10), while a range of values from individual sites in Europe, many monitored over the long term, are shown in Table 11. These data indicate that, although generally beneficial in terms of increased soil C, the effects of a switch from conventional to zero tillage are highly variable. Based on a number of long-term zero tillage studies in Europe, Smith *et al.* (1998) estimate that zero tillage can enhance the existing stock of SOC by 0.73 % per year, with an error (95 % CI) of around 50 % around this value, and, therefore, the measure could sequester around 50 Tg C yr<sup>-1</sup> (180 Tg CO<sub>2</sub>-eq yr<sup>-1</sup>) in Europe, while Lu *et al.* (2009) estimate potential for 17 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> to be sequestered by further uptake of no-till management in China. As

**Table 10.** Ranges of C sequestration rates from around the world in response to conversion from conventional to zero tillage (adapted from Desjardins *et al.*, 2005).

Region	C sequestration (t CO <sub>2</sub> ha <sup>-1</sup> yr <sup>-1</sup> )	Reference
Canada	0.73-1.28	McConkey <i>et al.</i> , 1999
USA	0.51-2.05	Lal, 1997
USA	1.83-2.93	Lal <i>et al.</i> , 1998
Developing countries	0.62-2.05	Lal, 1997
Asia	0.44-1.06	Lal, 1997
Africa	0.33-1.06	Lal, 1997
Latin America	0.44-1.06	Lal, 1997

**Table 11.** Reported effects of no-till on soil C storage compared to conventional tillage (adapted from Smith *et al.*, 1998).

Location and duration	Change in C storage (% SOC yr <sup>-1</sup> )	Reference
UK, 2 years	0.83	Tomlinson, 1974; Cannell and Finney, 1973
UK, 5 years	-0.19	Powlson and Jenkinson, 1981
UK, 6 years	0.99	Powlson and Jenkinson, 1981
UK, 8 years	0.48	Powlson and Jenkinson, 1981
UK, 8 years	1.15	Chaney <i>et al.</i> , 1985
UK, 10 years	-0.15	Powlson and Jenkinson, 1981
UK, 23 years	0.41	Ball <i>et al.</i> , 1994
Germany, 4 years	-0.58 – 1.52	Buhtz, <i>et al.</i> , 1970
Germany, 5 years	1.43 – 2.12	Fleige and Baeumer, 1974
Germany, 6 years	0.32	Fleige and Baeumer, 1974

discussed in section 4.2.3, some researchers argue that increases in soil C due to this practice have been over-estimated because shallow level soil sampling misses higher concentrations of SOC in the subsoil under conventional tillage (Carter, 2005; Dolan *et al.*, 2006; Baker *et al.*, 2007). Nevertheless, a meta-analysis excluding such studies shows a small overall increase in SOC on average due to reduced/zero tillage, though results are variable and give large uncertainty bounds (Ogle *et al.*, 2005). Small additional savings in CO<sub>2</sub> emissions are also made by reduced fossil fuel use; less diesel is used in ploughing, though slightly more in the production of additional herbicide necessary on zero till fields (see section 4.10).

Benefits in terms of N<sub>2</sub>O emission mitigation are questionable. However, because N<sub>2</sub>O is produced by both aerobic and anaerobic processes, a trade-off can occur, as short-term emissions from tillage stimulation of decomposition stop, but denitrification may increase as soil is less aerated and more compacted. Data shown in Table 12 illustrate the

wide range of N<sub>2</sub>O emission changes reported in the literature, ranging from a 280 % increase in emissions to a 52 % decrease. Smith *et al.* (2001) consider that as much as half of the mitigation potential could be lost through increases in N<sub>2</sub>O emissions as a result of increased anaerobicity of soils under zero tillage, and Six *et al.* (2004) argue that improved N management is required to maximize benefits from this form of management.

A modelling study using DNDC calculated that zero tillage management of maize production would increase N<sub>2</sub>O emissions by 4.6 kg N ha<sup>-1</sup> yr<sup>-1</sup> (2.1 t CO<sub>2</sub>-eq ha<sup>-1</sup> yr<sup>-1</sup>), offsetting the benefits in terms of increased C storage and reducing the total mitigation potential of the strategy by 75 % (Li *et al.*, 2005). Literature reviews by Gregorich *et al.* (2005) and Liebig *et al.* (2005) found no consistent relationship between tillage and N<sub>2</sub>O emission levels, with both increased and decreased emissions when no-till was compared to conventional tillage, and other researchers have reported that N<sub>2</sub>O emission from no-

**Table 12.** Reported effects of no-till soil N<sub>2</sub>O emissions in terms of percentage increase or decrease in comparison with conventional tillage treatment (partially adapted from Gregorich *et al.*, 2005).

Location and duration	% Change in N <sub>2</sub> O emissions, mean (range)	Reference
Canada, 4 years	-14.5 (-35 to 4.5)	Malhi and Lemke, 2007
Canada, 2 years	-51 (-52 to -50)	Malhi <i>et al.</i> , 2006
Denmark, 91 days	-25	Chatskikh and Olesen, 2007
Canada, 3 years	13 (-25 to 63)	Gregorich <i>et al.</i> , 2005
Canada, 3 years	145 (98 to 220)	Gregorich <i>et al.</i> , 2005
Canada, 2 years	-27 (-31 to -24)	Gregorich <i>et al.</i> , 2005
Canada, 2 years	23 (-27 to 49)	Gregorich <i>et al.</i> , 2005
Canada, 1 year	-14	MacKenzie <i>et al.</i> , 1998
Canada, 1 year	-37	MacKenzie <i>et al.</i> , 1998
Canada, 1 year	60	Gregorich <i>et al.</i> , 2005
Canada, 2 years	-15 (-26 to -3)	Kaharabata <i>et al.</i> , 2003
USA, 1 year	-65	Elder and Lal, 2008
Canada, 2 years	87.5	Mkhabela <i>et al.</i> , 2008
Scotland, 12 weeks	280	Ball <i>et al.</i> , 2008

till is not necessarily greater than from conventional tillage (Parkin and Kaspar, 2006; Venterea *et al.*, 2005). A study into the possible effects of no-till agriculture in the federal state of Baden-Wurttemberg in Germany, concluded that change to no-till on 40 % of cropland would reduce total GHG emissions from agriculture in the region by 5-14 % over 20 years, and emphasizes the need for the management strategy to be continued in the long term to prevent sequestered C being released again as soon as the land is tilled (Neufeldt, 2005).

Time is also an important factor in evaluating the impact of this measure, as a modelling study by Six *et al.* (2004) suggests that, for humid sites, no-till may increase N<sub>2</sub>O emissions for 10 years or more, but change to reducing emissions after 20 years. A review by Johnson *et al.* (2007), concluded that while N<sub>2</sub>O emissions may in general be increased by reduced tillage, there appears to be no consistent relationship, and the interaction of soil, climate and management system needs further evaluation. More recently, Rochette (2008) reviewed 25 field studies and observed that no-till generally increased N<sub>2</sub>O emissions in soils with poor aeration, but was generally neutral in soils with good and medium aeration. They concluded that increased N<sub>2</sub>O emissions may result in a negative GHG balance on many poorly-drained, fine-textured agricultural soils under no-till, in regions with a humid climate. Impacts on C sequestration may also be time-dependent as data from the USA suggests that net soil C gains may decline to zero after the initial

20 years of implementation, leaving only the indirect savings in terms of reduced fuel usage (Marland *et al.*, 2003).

#### 4.7.2 Reduced, conservation or minimum tillage

Studies into the effects of reduced tillage, and resulting effects on soil C storage, vary in the precise regime tested (Table 13). Meta-analysis of a global data set suggests that SOC will increase by 3-6 % (Ogle *et al.*, 2005). In France, reduced tillage has been reported to increase soil C stocks by 0.77 t CO<sub>2</sub> ha<sup>-1</sup> yr<sup>-1</sup> (Arrouays *et al.*, 2002), while Desjardins *et al.* (2005) cite data indicating that 0.22-2.2 t CO<sub>2</sub> ha<sup>-1</sup> yr<sup>-1</sup> has been sequestered by the adoption of conservation tillage in North America. In Bulgaria, stopping autumn ploughing and instead using a system of winter cutting and spring chisel cultivation, reduced losses of organic C from the soil by 3.5 times compared with traditional methods of maize cultivation (Dimitrov *et al.*, 2004). Again, these sequestration rates may over-estimate increases in C storage under shallower ploughing techniques in comparison with conventional tillage if they are only based on measurements taken in the top 20 or 30 cm of the soil profile (Baker *et al.*, 2007).

Shallower ploughing techniques have also been shown to reduce N<sub>2</sub>O emissions in comparison with conventional tillage; van der Weerden *et al.* (1999) reported that rotovation stimulated a three times increase in cumulative flux from a clover sward, while



traditional ploughing caused a five times increase. Chatskikh and Olesen (2007) reported that, over the course of a 91-day trial, reduced tillage led to a 21 % decrease in N<sub>2</sub>O emissions compared with conventional tillage, while Drury *et al.* (2006) report a 38 % decrease in the three-year average of N<sub>2</sub>O emissions from a wheat-corn-soybean rotation under a conservation tillage system. Other researchers report higher N<sub>2</sub>O emissions from reduced or conservation tillage systems (Beheydt *et al.*, 2008; D’Haene *et al.*, 2008; Koga *et al.*, 2004), differences that are not statistically significant (D’Haene *et al.*, 2008; Elmi *et al.*, 2003; ), or strong interactions with fertilizer type or placement (Venterea and Stanenas; 2008; Venterea *et al.*, 2005). Therefore, it is unclear whether any savings in terms of increased soil C storage will be significantly offset by increases in N<sub>2</sub>O emissions; results will depend on both environmental conditions and the exact regime chosen. A modelling study for the EU-25 calculated that a switch to reduced tillage would sequester 0.4 t CO<sub>2</sub> ha<sup>-1</sup> yr<sup>-1</sup>, while 0.7 t CO<sub>2</sub>

ha<sup>-1</sup> yr<sup>-1</sup> would be sequestered by a minimum tillage system, and argued that N<sub>2</sub>O emissions would be reduced on average, although, on a local level, result would not be uniform, exhibiting both increases and decreases in emissions (INSEA, 2006).

#### 4.7.3 Residue management

Residue management usually goes hand-in-hand with reduced or zero tillage. Residue incorporation, where stubble, straw or other crop debris is left on the field, and then incorporated when the field is tilled, is used in some areas for water conservation, but also enhances C returns to the soil, thereby encouraging C sequestration. Lu *et al.* (2009) argue that straw return could sequester 126 Tg CO<sub>2</sub> yr<sup>-1</sup> in China, representing the mitigation of 5.3 % of the country’s CO<sub>2</sub> emissions from fossil fuel emissions in 1990. However, incorporation can increase N<sub>2</sub>O emissions. For example, Gregorich *et al.* (2005) review data that indicate that incorporating stubble residues by ploughing in the autumn leads to double the N<sub>2</sub>O

**Table 13.** Reported effects of reduced or conservation tillage on soil C storage compared to conventional tillage (adapted from Alvarez, 2005).

Measure /treatment	Location and duration	Change in C storage (t CO <sub>2</sub> ha <sup>-1</sup> )	Reference
Conserv. till	Argentina, 6 years	31.2	Diaz-Zorita, 1999
Conserv. till	Argentina, 6 years	20.5	Krüger, 1996
Reduced till	Argentina, 6 years	18.3	Diaz-Zorita, 1999
Reduced till	Argentina, 6 years	4.4	Krüger, 1996
Conserv. till	Canada, 3-8 years	-23.5 – -2.6	Angers <i>et al.</i> , 1997
Conserv. till	Canada, 4 years	16.9 – 26.4	Franzluebbbers and Arshad, 1996a
Conserv. till	Canada, 4 years	13.6	Grant and Lafond, 1994
Conserv. till	Canada, 6 years	9.9	Franzluebbbers and Arshad, 1996b
Conserv. till	Canada, 7 years	-2.6 – 3.7	Franzluebbbers and Arshad, 1996a
Conserv. till	Canada, 16 years	-21.3 – -12.1	Franzluebbbers and Arshad, 1996a
Reduced till	Canada, 3 years	33	Angers <i>et al.</i> , 1997
Reduced till	Canada, 4 years	0.7	Grant and Lafond, 1994
Reduced till	Canada, 3-11 years	2.9 – 11	Campbell <i>et al.</i> , 1998
Conserv. till	Spain, 13 years	-4.8 – 5.9	Hernanz <i>et al.</i> , 2002
Reduced till	Spain, 13 years	13.9 – 17.2	Hernanz <i>et al.</i> , 2002
Conserv. till	USA, 11 years	12.8	Yang and Wander, 1999
Conserv. till	USA, 12 years	-0.4 – 17.6	Halvorson <i>et al.</i> , 2002
Reduced till	USA, 3 years	-25.7 – -11.4	Dao <i>et al.</i> , 2002
Reduced till	USA, 8.5 years	9.2	Yang and Wander, 1999
Reduced till	USA, 11 years	-9.5	Yang and Wander, 1999
Reduced till	USA, 12 years	-9.5 – 6.2	Halvorson <i>et al.</i> , 2002

Note: Conserv. = conservation

emissions of leaving the residues on the surface over winter. Net benefits in terms of climate change mitigation may be highest when residues with high N content are removed. Composting these residues and then returning them to the soil may reduce in-situ N<sub>2</sub>O emissions compared with incorporating untreated residues, while retaining benefits in terms of reduced requirements for mineral fertilizer (Velthof and Kuikman, 2000). However, GHG emissions occur during composting, and active aeration or turning also contributes to indirect emissions from fuel use; in terms of total GWP, these may well outweigh any savings from direct N<sub>2</sub>O emissions (Hao *et al.*, 2001 and 2004). Baggs *et al.* (2003) highlight that residue management interacts with both tillage and fertilizer systems, reporting the results of field trials where surface mulching of residues in no-till treatments led to greater N<sub>2</sub>O losses from applied fertilizer than incorporating residues.

A modelling study using DNDC calculated that increasing residue incorporation from 15 to 90 % for a maize-wheat system in China would sequester 680 kg C ha<sup>-1</sup> yr<sup>-1</sup> as SOC, but that N<sub>2</sub>O emissions would rise sufficiently to give a net increase of 140 kg CO<sub>2</sub>-eq ha<sup>-1</sup> yr<sup>-1</sup> (Li *et al.*, 2005). However, Smith *et al.* (2000c) argue that the incorporation of cereal straw across Europe would have a net positive effect with increased N<sub>2</sub>O emissions being outweighed by the increases in SOC storage, and have also calculated that incorporation of 100 % of cereal straw in the UK would be a similarly positive step (Smith *et al.*, 2000a,b).

## 4.8 Mitigation potential of agronomy measures

As shown in Table 6, Smith *et al.* (2007b) have calculated that agronomy measures in cool dry climates could mitigate total GHGs by 0.39 (0.07-0.71) t CO<sub>2</sub>-eq ha<sup>-1</sup> yr<sup>-1</sup>, comprising an increase in SOC of 0.29 (0.07-0.51) t CO<sub>2</sub>-eq ha<sup>-1</sup> yr<sup>-1</sup>, and decreased N<sub>2</sub>O emissions of 0.1 (0.0-0.2) t CO<sub>2</sub>-eq ha<sup>-1</sup> yr<sup>-1</sup>. In cool moist climates, the figures were 0.98 (0.51-1.45) t CO<sub>2</sub>-eq ha<sup>-1</sup> yr<sup>-1</sup>, comprising an increase in SOC of 0.88 (0.51-1.25) t CO<sub>2</sub>-eq ha<sup>-1</sup> yr<sup>-1</sup>, and decreased N<sub>2</sub>O emissions of 0.1 (0.0-0.2) t CO<sub>2</sub>-eq ha<sup>-1</sup> yr<sup>-1</sup>. Many other researchers have examined individual measures within this category, and a range of studies are reviewed in the following sections.

### 4.8.1 Catch crops and increased cover

Meta-analysis of studies from a range of countries, covering different climatic and agricultural systems, calculated increases in SOC of around 7-11 % over 20 years in response to the use of organic amendments (Ogle *et al.*, 2005). In France, catch crops have

been reported to increase SOC by the equivalent of 0.55 t CO<sub>2</sub> ha<sup>-1</sup> yr<sup>-1</sup> (Arrouays *et al.*, 2002), while studies carried out in North America on the effect of reducing summer fallow indicate that C sequestration can vary from 0.07 to 1.47 t CO<sub>2</sub> ha<sup>-1</sup> yr<sup>-1</sup> (Campbell *et al.*, 2001; Lal, 2001; Smith *et al.*, 2001; McConkey *et al.*, 1999). Bulgarian studies carried out in test plots have reported that winter cover vastly reduces (by as much as 500 times) losses of SOC and nutrients in comparison with fallow periods (Petrova, 1989; Tsvetkova *et al.*, 1995; Mihailova *et al.*, 2001). In terms of N<sub>2</sub>O emissions, winter cover has been reported to reduce % N<sub>2</sub>O emission factors by around 0.25 in the Netherlands (Velthof and Kuikman, 2000).

### 4.8.2 Crop selection and rotation

In their review of management strategies to sequester C in agricultural soils, Desjardins *et al.* (2005) cite data indicating that including more forage crops, especially legumes, can sequester between 1.6 and 2.8 t CO<sub>2</sub> ha<sup>-1</sup> yr<sup>-1</sup>. N fixing crops can increase N<sub>2</sub>O emissions though. Mosier *et al.* (2006) report that the addition of a soybean rotation increased background N<sub>2</sub>O emissions from no-till continuous corn by 300 %. Rochette *et al.* (2004) report that emissions from unfertilized non-leguminous crops are generally lower than those from an actively growing legume crop, but Parkin and Kasper (2006) reported higher emissions from soils planted to fertilized corn than those planted to soybean. This is because biologically-fixed N is generally less available than mineral N from fertilizers, while the legume crop is actively growing, and illustrates that when N<sub>2</sub>O emissions from fertilizer N are taken into consideration, total N<sub>2</sub>O emissions can be lowered by the addition of N fixing crops. Meyer-Aurich *et al.* (2006) argue that the reduced fertilizer requirement of corn grown in rotation with red clover mitigates around 250 kg CO<sub>2</sub>-eq ha<sup>-1</sup>, and more than offsets the increased N<sub>2</sub>O emissions from the clover crop. Gregorich *et al.* (2005) reviewed studies carried out in Canada and concluded that emissions from legumes amount to 0.98 ± 0.58 t CO<sub>2</sub>-eq ha<sup>-1</sup> yr<sup>-1</sup>, suggesting that, at the bottom of this range, N fertilizer applications of just 86 kg ha<sup>-1</sup> yr<sup>-1</sup> would give equivalent emissions, not including those from the fertilizer manufacture (assuming a 1 % EF).

According to Snyder *et al.* (2009), few studies have examined the long-term effects of different cropping systems on net GHG emissions, but those that have include Adviento-Borbe *et al.* (2007), who found that the GWP of high-yielding continuous maize was 2-3 times higher than that of the conventionally-tilled maize-wheat-soybean rotation studied by Robertson *et al.* (2000). When compared on the basis of GWP (in CO<sub>2</sub>-eq) per unit of food or food energy produced, the high-yielding, intensively-managed continuous maize

studied by Adviento-Borbe *et al.* (2007) gave a ratio of 60 kg CO<sub>2</sub>-eq per Gcal of food yield. This compared to ratios of 95 for the conventional till, 10.8 for the no-till, and 52.5 for the “low-input with legume cover crop” treatments in the maize-wheat-soybean rotation studied by Robertson *et al.* (2000). The factor causing the greatest GWP difference in these two studies was the impact of no-till management on estimated (to a depth of 7.5 cm) net soil C storage (Snyder *et al.*, 2009), and, therefore, the differences may not have been as significant if deeper soil sampling had been employed.

## 4.9 Mitigation potential in rice production

### 4.9.1 Water management

Water management, e.g. reducing the time during which the fields are flooded, can reduce GHG emissions by about 50 % (Cole *et al.*, 1997). For example, in China, draining the field intermittently or, even better, during the second half of the growing season, has been shown to substantially reduce CH<sub>4</sub> emissions and increase production (FAO, 1994). Majumdar (2003) suggests that intermittent drainage is the most effective CH<sub>4</sub> emission mitigation practice for irrigated rice paddies, as it has mitigated up to 80 % of CH<sub>4</sub> emissions, although it can be as low as 7 %, depending on the timing, frequency and span of drainage (Wassman *et al.*, 2000a) (Table 14). Taking into account the level of implementation that would be practically achievable, Kern *et al.* (1997) estimate that intermittent drainage could reduce CH<sub>4</sub> emissions from Chinese rice paddies by 10 %

(compared to 1990 levels), assuming the practice was adopted on 33 % of the paddy area. This indicates the importance of taking practical feasibility into account when estimating mitigation potential at a larger scale.

In India, the type of water regime used to grow rice has been shown to strongly influence CH<sub>4</sub> emission levels. Gupta *et al.* (2009) report that national annual CH<sub>4</sub> emission factors have been estimated as 17.48 ± 4 g m<sup>-2</sup> for irrigated continuously flooded paddies, 6.95 ± 1.86 g m<sup>-2</sup> for rainfed drought-prone paddies, 19 ± 6 g m<sup>-2</sup> for rainfed flood-prone and deep-water paddies, 6.62 ± 1.89 g m<sup>-2</sup> for irrigated intermittently flooded single aeration managed paddies, and 2.01 ± 1.49 g m<sup>-2</sup> for irrigated intermittently flooded multiple aeration paddy water regimes. Yan *et al.* (2003) report a similar pattern of EFs for rice growing regions across Asia. These data also support the argument that intermittent drainage reduces emissions, as the highest EFs are for paddies that are most frequently flooded.

The benefits of reduced CH<sub>4</sub> emissions with intermittent drainage of flooded fields, which directly impacts nitrification and denitrification rates, should be carefully weighed against the risks of increased N<sub>2</sub>O emissions (Guo and Zhou, 2007). For example, in China, where manure and urea were the N sources, CH<sub>4</sub> emissions occurred at lower redox (more reduced) potentials (< -100 mV) than did N<sub>2</sub>O emissions (> +200 mV), and there was a significant inverse relationship between emissions of these two GHGs (Hou, *et al.*, 2000). Maintaining the soil redox potential between -100 and +200 mV was cited as preventing CH<sub>4</sub> production, and would also be low enough to encourage N<sub>2</sub>O reduction to N<sub>2</sub>.

**Table 14.** Effectiveness of water management based strategies for mitigating CH<sub>4</sub> emissions from rice paddies (partially adapted from Majumdar, 2003).

Mitigation practice	CH <sub>4</sub> emission reduction	Reference
Mid-season drainage	60 %	Bronson <i>et al.</i> , 1997
	43 %	Corton <i>et al.</i> , 2000
	44 %	Lu <i>et al.</i> , 2000
	23 %	Wang <i>et al.</i> , 2000
	7-80 %	Wassman <i>et al.</i> , 2000a
Intermittent irrigation	36 %	Shin <i>et al.</i> , 1996
	61 %	Lu <i>et al.</i> , 2000
Seasonal drainage	63-72 %	Cai <i>et al.</i> , 1998
	42-67 %	Wassman <i>et al.</i> , 2000c
Winter drainage	15-79 %	Kang <i>et al.</i> , 2002
Mid-tillering drainage	15-80 %	Wassman <i>et al.</i> , 2000b

#### 4.9.2 Rice cultivar

Improved rice cultivars have been estimated to reduce GHG emissions by up to 20 % (Sass *et al.*, 1992). Studies have shown that CH<sub>4</sub> emissions can differ significantly between rice cultivars due to differences in CH<sub>4</sub> production, oxidation and transport capacities. For example, Wang *et al.* (1997) reported that emissions could be reduced by around 60 % by cultivar choice (comparing varieties in the same growth stage). Watanabe *et al.* (1995) reported an approximately 30 % difference in emissions between the highest and lowest emitting cultivars tested over a two-month period, and Mitre *et al.* (1999) found that seasonal CH<sub>4</sub> emissions could be reduced by a maximum of 43 % by cultivar selection. Most recently, Lou *et al.* (2008) found that average daily CH<sub>4</sub> emissions were 48 % lower from the lowest emitting cultivar tested, in comparison with the variety that emitted the most CH<sub>4</sub>.

Rice plants can influence CH<sub>4</sub> production by enhancing anaerobic conditions due to root respiration, and by providing substrates for methanogens, such as root exudates (Neue *et al.*, 1997; Wang *et al.*, 1997). Different rice cultivars have shown different patterns, amounts and chemical compositions of root exudates (Ladha *et al.*, 1986; Lin and You, 1989). Aulakh *et al.* (2001) argue that CH<sub>4</sub> production and emission are more closely related to the release pattern of root exudate-C than to its individual components and, therefore, the cultivars with the lowest excretion rate represent the best choice in terms of mitigation potential. The plants can also affect CH<sub>4</sub> oxidation by enzymatic oxidation and by diffusion of oxygen through aerenchyma into the rhizosphere (Epp and Chanton, 1993). Wang *et al.* (1997) report that CH<sub>4</sub> oxidation, evaluated using root airspace and root oxidation power, differs significantly among rice cultivars. Finally, plant-mediated transport has been shown to account for more than 90 % of the total emissions (Inubushi *et al.*, 1989; Sass *et al.*, 1990), and differs between rice varieties due to the differences in tiller number and aerenchyma system (Butterbach-Bahl *et al.*, 1997; Wang *et al.*, 1997; Inubushi *et al.*, 2003).

#### 4.9.3 Fertilization and other additions

Improved fertilization has been estimated to reduce GHG emissions from rice paddies by 28–36 % (Lindau *et al.*, 1993). Applications of phosphate fertilizer reduce CH<sub>4</sub> emissions, as it decreases the total amount of root exudation by decreasing the root/shoot ratio (Lu *et al.*, 1999), and also because methanogens that colonize the root surface are sensitive to phosphate (Conrad *et al.*, 2000). Nitrogen fertilizers may contribute to the mitigation of CH<sub>4</sub> emissions because nitrate, like sulphate (SO<sub>4</sub><sup>2-</sup>) and ferric iron (Fe<sup>3+</sup>), inhibits the production of CH<sub>4</sub>, as nitrate reducers

utilize acetate and hydrogen more efficiently than methanogens (Acht nich *et al.*, 1995; Klüber and Conrad, 1998a; Chidthaisong and Conrad, 2000), and because nitrite, nitrogen oxide (NO) and N<sub>2</sub>O, which accumulate transiently during the reduction of nitrate, are toxic to methanogens (Klüber and Conrad, 1998b). N fertilization also stimulates the CH<sub>4</sub> oxidizing bacteria in the rhizosphere, further reducing emissions (Bodelier *et al.*, 2000a, b). However, excess N can result in increased N<sub>2</sub>O emissions, so care must be taken to avoid a trade-off in different GHG emissions. A review by Majumdar (2003) considers measures that mitigate both gases and suggests the use of nitrification or urease inhibitors and slow-release fertilizers, which, as shown in Table 15, have been reported to reduce both CH<sub>4</sub> and N<sub>2</sub>O emissions from rice paddies. However, these agrochemicals are not commonly used by rice growers and may be a costly option compared to concentrating solely on measures to mitigate CH<sub>4</sub> emissions, especially as N<sub>2</sub>O emissions from rice paddies are much lower, even when their increased radiative forcing is taken into account (Ghosh, 1998).

A further mitigation option is fertilization with iron (Conrad, 2002). Increased ferric iron content in the rhizosphere seems to suppress CH<sub>4</sub> formation (Watanabe and Kimura, 1999; Jäckel and Schnell, 2000), and field experiments with iron (ferrihydrite) fertilization have shown a 50 % decrease in CH<sub>4</sub> emissions without compromising crop yields (Jäckel *et al.*, 2005) (Table 15). Other additions that have been suggested include sulphate, which, as mentioned above, inhibits CH<sub>4</sub> production (see Table 15, and a review by Denier van der Gon *et al.*, 2001, which discusses mechanisms, mitigation potential and costs), and potassium. Babu *et al.* (2006) report that application of 30 kg K ha<sup>-1</sup> as potassium chloride, reduced CH<sub>4</sub> emissions from a flooded soil planted with rice by 49 %, by preventing a drop in the redox potential and thereby making less active reducing substances and ferrous iron (Fe<sup>2+</sup>) available in the rhizosphere, and also by inhibiting methanogenic bacteria and stimulating the methanotrophic bacterial population.

#### 4.9.4 Treatment of crop residues

Many researchers have reported that organic amendments such as rice straw stimulate CH<sub>4</sub> production and increase emissions from rice paddies (Schütz *et al.*, 1989; Yagi and Minami, 1990; Sass *et al.*, 1991; Denier van der Gon and Neue, 1995; Chidthaisong *et al.*, 1996; Rath *et al.*, 1999; Setyanto *et al.*, 2000). However, burning residues releases GHG and can also cause serious air quality problems, resulting in the practice being banned by many Asian governments (Singh *et al.*, 2008). Therefore, alternatives such as composting the residues prior to

**Table 15.** Effectiveness of nitrification or urease inhibitors, slow-release fertilizers and other additions, for mitigating CH<sub>4</sub> and N<sub>2</sub>O emissions from irrigated rice paddies (adapted from Majumdar, 2003).

Addition type / mitigation practice	CH <sub>4</sub> emission reduction	N <sub>2</sub> O emission reduction	Reference
<b>Nitrification inhibitors</b>			
Encapsulated calcium carbide	90 %	44 %	Bronson and Mosier, 1991
Urea + calcium carbide	36 %		Lindau <i>et al.</i> , 1993
Urea + DCD	14 %		Lindau <i>et al.</i> , 1993
	22 %	53 %	Ghosh, 1998
		11 %	Kumar <i>et al.</i> , 2000
Ammonium sulphate + thiosulphate		9 %	Kumar <i>et al.</i> , 2000
Ammonium sulphate + DCD	13 %	46 %	Ghosh, 1998
		26 %	Kumar <i>et al.</i> , 2000
<b>Urease and nitrification inhibitors</b>			
Urea + hydroquinone + DCD	66 %	50 %	Zhou <i>et al.</i> , 1999
<b>Slow-release N</b>			
Tablet urea	10-39 %		Wassman <i>et al.</i> , 2000b
Wax-coated calcium carbide		73 %	Mosier <i>et al.</i> , 1994
Polyolefin coating		80 %	Shoji and Kanno, 1994
<b>Sulphate addition</b>			
Sodium sulphate	28-35 %		Lindau <i>et al.</i> , 1993
Gypsum	55-70 %		Denier van der Gon and Neue, 1994
Ammonium sulphate	58 %		Shao and Li, 1997
	42-60 %		Cai <i>et al.</i> , 1997
	25-36 %		Corton <i>et al.</i> , 2000
	10-67 %		Wassman <i>et al.</i> , 2000a
Phosphogypsum	9-73 %		Wassman <i>et al.</i> , 2000a
	72 %		Corton <i>et al.</i> , 2000
		100 %	Lindau <i>et al.</i> , 1998
<b>Iron addition</b>			
Fe(OH) <sub>3</sub>	46 %		Watanabe and Kimura, 1999
Ferrihydride 15 g kg <sup>-1</sup> soil 30 g kg <sup>-1</sup> soil	43 %		Jäckel and Schell, 2000
	84 %		Jäckel and Schell, 2000

field application have been suggested as mitigation practices (Yagi and Minami, 1990; Minami and Neue, 1994). Shin *et al.* (1996) reported that the incorporation of well decomposed (composted) rice straw reduced CH<sub>4</sub> emissions by as much as 49 % compared to fresh rice straw amendments. Other researchers suggest that rice straw should be removed (Liou *et al.*, 2003), without adverse affects on soil organic matter (Singh *et al.*, 2008), providing residues for use on other crops, or to generate energy. Adjusting the timing of organic residue additions so that they are incorporated during the dry period rather than during flooded periods may also help to reduce emissions (Xu *et al.*, 2000; Cai and Xu, 2004).

## 4.10 Mitigation of indirect emissions

### 4.10.1 Emissions from fertilizer production

Although any reductions in global fertilizer consumption will obviously have a knock-on effect on total emissions from its production, it is also possible to reduce GHG emissions per unit of fertilizer produced by improving the efficiency of production plants. According to Al Ansari (2007), even just universal implementation of the current best available technology would save 40 % of current energy use and reduce GHG emissions by 58 %. Technological advances are continuing to try and improve this



even further, e.g. via the use of C capture technology (Audus, 2007) for CO<sub>2</sub> from ammonia plants. For nitric acid plants, Jenssen (2007) suggests that the emissions of N<sub>2</sub>O could be reduced by 70-90 % in most plants by installing modern N<sub>2</sub>O abatement technology. In practice, if global incentive systems were adopted for installing new technologies and for making energy efficiency improvements, it is likely that the global emissions from the fertilizer industry could be reduced by more than 30 %, saving some 200 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> (IFA, 2009a).

#### 4.10.2 On-farm energy use

According to Smith *et al.* (2007a), mitigation of 770 Tg CO<sub>2</sub>-eq yr<sup>-1</sup> could be achieved by 2030 by improved energy efficiency in agriculture. An earlier IPCC assessment suggested that a 10-40 % reduction in fuel use was achievable via reductions in tillage, better fertilizer efficiency, improved irrigation techniques and more use of solar drying (Cole *et al.*, 1996). Since tillage and harvesting consume the most fuel within

agricultural systems (Frye, 1984), encouraging no-till or reduced tillage practices, as well as possible impacts on soil C storage and N<sub>2</sub>O emissions (see section 4.7), will also reduce CO<sub>2</sub> emissions indirectly via reduced energy consumption (Archer *et al.*, 2002). According to Frye (1984), shifting from moldboard ploughing to no-till could save about 70 kg CO<sub>2</sub>-eq ha<sup>-1</sup> yr<sup>-1</sup>. This practice can yield significant results. In Australia, fuel use by the cereals industry has roughly halved between 1990 and 2004, and this is at least partly due to an increase in no-till management from 2 Mha (~40 % of total area under cereals) in 1996, to just under 5 Mha (~70 %) in 2004 (Ugalde *et al.*, 2007). In the USA, adoption of reduced tillage practices from 1990 to 2004 has been reported to have reduced annual net fossil fuel emissions by 8.8 Tg CO<sub>2</sub>-eq (Nelson *et al.*, 2009). Even just switching to shallower tilling can reduce energy requirements. Meyer-Aurich *et al.* (2006) report that replacing a moldboard plough with a chisel plough mitigates 39 kg CO<sub>2</sub>-eq ha<sup>-1</sup>.

## 5. Conclusions

Agriculture contributes a significant proportion of global GHG emissions, and these emissions are predicted to increase as the global population rises. With predictions that 9 billion people will need to be fed by the middle of this century, agricultural production will need to increase, but the additional pressure on land availability means that improving the productivity of existing agricultural land is the best way to achieve this. Using the existing cropland most efficiently may also contribute to preserving areas with valuable biodiversity and high carbon sequestration potential against land use changes. There is much scope for more efficient use of resources, particularly N, within the current level of understanding. The wider implementation of BMP strategies can play a significant role in limiting the impact of agricultural production on the future climate.

Currently, NUE (measured by crop production per unit of N applied) is falling in many countries, such as China, where some cropping systems are over-fertilized, while soils in other regions, such as parts of Africa and India, still suffer from chronic nutrient deficiency. This imbalance needs to be addressed, and better integration of organic nutrient sources such as animal waste and crop residues into crop nutrition programmes can assist in this while also helping to mitigate indirect emissions from fertilizer production.

Increased use of organic material for fertilization will also improve soil quality by increasing SOC. Overall, cropland management mitigation strategies, such as further implementation of fertilizer BMPs, offer the highest potential for mitigating future agricultural GHG emissions. Fertilizer production also has a significant role to play, as the most advanced technology available today can significantly reduce the GHG emissions associated with the older, less efficient plants still used in many parts of the world.

Many uncertainties remain regarding the exact potential of various mitigation practices, not least because C and N dynamics in agricultural systems can be very variable, both between different sites and cropping systems and also within the same area, over time or with depth in the soil profile. This makes generalisations difficult. This study has identified multiple data gaps, not least the paucity of data relating to tropical agricultural systems in Latin America, Southeast Asia and parts of Africa. However, there is more than sufficient understanding of general impacts to support strong action on the part of the international community. The technology already exists to ensure croplands can play a significant role in mitigating climate change while still meeting the demand of feeding an increasing global population.

## Appendix A – Comparing methodologies for calculating N<sub>2</sub>O emissions

**Table A1.** A range of reported N<sub>2</sub>O emission factors (% of applied N emitted as N<sub>2</sub>O-N) based on field studies (partially adapted from Snyder *et al.*, 2009)

Fertilizer type, conditions, etc.	Emission factor, or range, % (number of observations)	Reference
IPCC default, mineral fertilizers	1	IPCC, 2006
IPCC default, all fertilizer incl. organic	1.25	IPCC, 1997
Anhydrous ammonia	0.05-19.6 (38)	Stehfest and Bouwman, 2006
Anhydrous ammonia	0.05-6.8	Granli and Bøckman, 1994
Anhydrous ammonia	0.04-6.84 (12)	Eichner, 1990
Ammonium carbonate, chloride or sulphate	0.01-36.54 (74)	Stehfest and Bouwman, 2006
Ammonium carbonate, chloride or sulphate	0.03-0.90 (11)	Eichner, 1990
Ammonium phosphates	0.06-7 (6)	Stehfest and Bouwman, 2006
Ammonium sulphate	0.08-0.18 (3)	Eichner, 1990
Ammonium-based	0.03-1.8	Granli and Bøckman, 1994
Ammonium-based	0.04-0.12 (4)	Eichner, 1990
Ammonium nitrate	0-30.4 (131)	Stehfest and Bouwman, 2006
Ammonium nitrate	0.04-1.7	Granli and Bøckman, 1994
Ammonium nitrate	0.04-1.71 (8)	Eichner, 1990
Calcium nitrate	0.01-1.75 (7)	Eichner, 1990
Calcium ammonium nitrate	0.05-11 (73)	Stehfest and Bouwman, 2006
Potassium nitrate	0.02 (1)	Eichner, 1990
Sodium nitrate	0-0.5 (9)	Eichner, 1990
Calcium nitrate, potassium nitrate, sodium nitrate	0-41.8 (58)	Stehfest and Bouwman, 2006
Nitrate-based	0.001-1.8	Granli and Bøckman, 1994
Urea ammonium nitrate	0.03-16.03 (40)	Stehfest and Bouwman, 2006
Urea ammonium nitrate	1.57 (1)	Eichner, 1990
Urea	0.01-46.44 (131)	Stehfest and Bouwman, 2006
Urea	0.01-2.1	Granli and Bøckman, 1994
Urea	0.08-0.18 (7)	Eichner, 1990
Mix of synthetic fertilizers	0-16.78 (45)	Stehfest and Bouwman, 2006
Ammonium nitrate, UK sites	0.4-6.5 (12)	Dobbie and Smith, 2003
Ammonium-based	0.5-3	Jacinthe and Dick, 1997
Calcium ammonium nitrate, plus P and K	0.8 ± 0.2	Hellebrand <i>et al.</i> , 2008
Organic-synthetic fertilizer mixes	0-31.73 (48)	Stehfest and Bouwman, 2006
Organic-synthetic fertilizer mixes	1.78-1.8 (2)	Eichner, 1990
Organic	0.03-56 (88)	Stehfest and Bouwman, 2006
Organic	0.01-2.05	Granli and Bøckman, 1994
Organic, Scotland	0.01-1.65	Akiyama <i>et al.</i> , 2004

As shown in Table A1, a wide range of emissions factors (EFs) have been reported and, therefore, when calculating emissions on a spatial basis at country level or below, many researchers have used specific EFs (a tier 2 methodology; IPCC, 2001), which, as well as being dependent on fertilizer type, can also be crop-specific and/or dependent on climatic conditions, tillage practice, or models of nutrient cycling (a tier 3 methodology; IPCC, 2001) to determine direct N<sub>2</sub>O emissions from fertilizer applications. Table A2 gives a range of fertilizer-induced N<sub>2</sub>O emissions from different sectors within countries, comparing calculations made using these methodologies with those using the IPCC default EFs.

Flynn *et al.* (2005) used crop specific and, in some cases, climate-dependent, EFs to calculate N<sub>2</sub>O emissions from fertilizer applied to Scottish agricultural land, and they argued that the IPCC default EFs over-estimated emissions from cereals and oilseed rape, but underestimated those from other arable crops, such as potatoes and leafy vegetables. This is because previous studies have indicated that mean N<sub>2</sub>O-N emissions from cereals and oilseed rape in the UK are around 0.6 % of applied mineral N and do not show a clear response to rainfall (Dobbie *et al.*, 1999; Smith and Dobbie, 2002; Dobbie and Smith, 2003), while vegetables, particularly potatoes, seem to show a climate-dependent response to mineral fertilizer additions, more in line with patterns of emissions shown by grassland systems (Dobbie *et al.*, 1999; Flessa *et al.*, 2002; Dobbie and Smith, 2003). Overall, they concluded that emissions from mineral N applications to arable crops in Scotland were 0.23 Tg CO<sub>2</sub>-eq yr<sup>-1</sup>, around 60 % of the level of emissions calculated using the IPCC default EF of 1.25 % (Flynn *et al.*, 2005). In contrast, Brown *et al.* (2002) used an adapted version of the DNDC model to calculate numerous specific EFs for combinations of UK crops and dominant soil types. For emissions directly from mineral fertilizer applications, these EFs ranged from 0.1–14.8 %, although the mean was 1 %, and over 90 % of the values were below 2 %. Again, the authors note that EFs for potatoes and other vegetables were much larger than for cereal crops. Using data from 1990, average UK N<sub>2</sub>O emissions from mineral fertilizer applications and applied animal waste for cereal crops were calculated as 0.56 t CO<sub>2</sub>-eq ha<sup>-1</sup> yr<sup>-1</sup> and those from other crops as 1.02 t CO<sub>2</sub>-eq ha<sup>-1</sup> yr<sup>-1</sup> (Brown *et al.*, 2002). Data for Scotland is shown in Table A2 for comparison with methodologies used by Flynn *et al.* (2005); differences are at least partially attributable to the difficulty in obtaining accurate data on manure application levels.

The DNDC model has also been used to assess soil N<sub>2</sub>O emissions from countries such as China, where fertilization levels are much higher in comparison with

European countries such as Scotland. Li *et al.* (2001) used DNDC to model county-scale emissions for crop/management scenarios taking into account local weather and soil conditions, producing an inventory of soil N<sub>2</sub>O emissions for croplands in mainland China, which was then compared to calculations made using the IPCC methodology. On a national level, the two methodologies gave similar total values, the IPCC methodology produced an estimate of 167 Tg CO<sub>2</sub>-eq for total direct soil N<sub>2</sub>O emissions for 1990, 16 % higher than the DNDC estimate of 144 Tg CO<sub>2</sub>-eq (see Table A2 for the breakdown of emissions by source), but at county-scale they varied widely. DNDC estimated emissions more than 1.5 times larger than the IPCC methodology for a number of counties, which is probably a significant level of disagreement even given the large uncertainty in the estimates, and over three times larger in some cases. This is because DNDC calculated EFs varying from less than 0.25 % (of applied N emitted as N<sub>2</sub>O) up to over 4 %, although the national average was 0.8 %.

A further investigation comparing the N<sub>2</sub>O inventory of China when calculated by different methods, highlights an important issue regarding the comparison of climate-dependant methodologies with the IPCC methodology, which does not take into account any climatic variables. Lu *et al.* (2006) used an empirical model relating soil N<sub>2</sub>O emissions to N inputs and precipitation, and reported emission estimates for 1997 and 1998 made using both this model and the IPCC methodology. For 1997, the IPCC methodology gave an estimate of 168 Tg CO<sub>2</sub>-eq, 20 % higher than the 136 Tg CO<sub>2</sub>-eq estimated using the model. However, for 1998, a 14 % increase in rainfall resulted in the IPCC estimate actually being lower than that calculated by the model, 171 Tg CO<sub>2</sub>-eq (increased due to increased N fertilizer applications) vs. 183 Tg CO<sub>2</sub>-eq, indicating that such methodologies are much more susceptible to inter-annual variation in emissions.

The IPCC default EF applies to both mineral and organic fertilizers and, like mineral fertilizers, a wide range of EFs have been reported from the application of manure or slurry (Table A2), leading some researchers to argue that organic additions generally lead to higher emissions (see e.g. Lowrance *et al.*, 1998; Velthof *et al.*, 2003). However, Gregorich *et al.* (2005) report that N<sub>2</sub>O emissions from solid manure applications can be around three times lower than for applied liquid manure, and Flynn *et al.* (2005) argue that a lower EF of only 0.5 % (in comparison with 0.6 % or higher for mineral fertilizers) is reasonable for spreader applied slurry and farmyard manure applied to Scottish agricultural land. This lies within the 0.01–1.65 % range of emissions reported for organic fertilizers applied to Scottish soils (Akiyama *et al.*, 2004), and

**Table A2.** A comparison of the use of specific EFs, modelling techniques, and IPCC default EFs to calculate N<sub>2</sub>O emissions (converted to CO<sub>2</sub>-eq) from fertilizer applications

Crop, fertilizer type and location	N <sub>2</sub> O emissions (Tg CO <sub>2</sub> -eq yr <sup>-1</sup> ) calculated using:			Reference
	Specific EFs / empirical methods	Process-based modelling	IPCC default <sup>a</sup>	
<b>Scotland</b>				
Cereals and oilseed rape, mineral fertilizers	0.17		0.35	Flynn <i>et al.</i> , 2005
Potatoes and other arable, mineral fertilizers	0.06		0.03	Flynn <i>et al.</i> , 2005
Cereals and oilseed rape, applied manure	0.08		0.20	Flynn <i>et al.</i> , 2005
Potatoes and other arable, applied manure	0.02		0.04	Flynn <i>et al.</i> , 2005
All crops including grass, mineral fertilizer		0.91		Brown <i>et al.</i> , 2002
Cereals, mineral N and applied manure		0.10		Brown <i>et al.</i> , 2002
Other crops, mineral N and applied manure		0.06		Brown <i>et al.</i> , 2002
<b>China</b>				
All crops, mineral fertilizer		60.47	98.68	Li <i>et al.</i> , 2001
All crops, applied manure		9.30	22.33	Li <i>et al.</i> , 2001
All direct soil N <sub>2</sub> O, 'dry' year	136		168	Lu <i>et al.</i> , 2006
All direct soil N <sub>2</sub> O, wetter year	183		171	Lu <i>et al.</i> , 2006

<sup>a</sup>1.25 % of applied N, as per IPCC (1997).

is based on comparisons between laboratory and field tests. Under laboratory conditions, N<sub>2</sub>O emissions of only 0.2 % of applied N have been reported from cattle manure and surface-applied slurry (Flessa and Beese, 2000; Akiyama *et al.*, 2004), but comparing the results for applied urea N with Scottish field studies suggests that these emissions may be closer to 0.5 % under field conditions (Akiyama *et al.*, 2004). These studies were carried out on grassland rather than cropland, but in the absence of studies on N<sub>2</sub>O emissions from applied organic N fertilizers on arable land, this EF was assumed to apply to all crop types. Flynn *et al.*

(2005) also did not treat emissions from organic N applications as climate variable, in accordance with Ball *et al.* (2004) who carried out field studies in Scotland, and reported that use of organic manures avoids the very high fluxes associated with heavy rainfall occurring around the time of application, and can therefore mitigate total N<sub>2</sub>O emissions over a growing season by approximately 90 % in comparison with mineral fertilizers, though studies elsewhere show higher N<sub>2</sub>O emissions upon animal slurry applications (Rochette *et al.*, 2000).



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