Greenhouse Gas Emissions from European Grasslands

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• Executive Summary

In the EU 15, there were about 55 million ha of grasslands in the year 2000. Since 1990 (the reference year in the Kyoto protocol) some 3 million ha of grassland were converted to arable land (-5 % within 10 years), especially for maize cultivation. Within the EU, grasslands are likely to be a net sink of carbon dioxide, but a source of nitrous oxide (from fertilisation and manuring of soils) and of methane (mainly from the enteric fermentation of ruminants). Since agricultural management is one of the key drivers of the sequestration and emission processes, there is potential within the EU to reduce the net GHG flux, expressed in CO_2 equivalents.

For national inventories of GHG budgets numerous methods are available but the most widely used are the IPCC 1996 revised guidelines. These provide default emission factors, but allow for country / region specific values of these factors if available. Other, more sophisticated methods may also be used if available. In particular, comparisons and aggregation of data obtained at different European sites could help to provide more relevant values for emission factors used in Europe today. In addition, the use of dynamic emission factors (that respond to, for example, climate, soils, crop, fertiliser etc.) might replace the static default emission factors currently used and improve their accuracy and flexibility. Well-evaluated process-based models, tested at a series of benchmark, may also play a role in GHG accounting in the future. However, verification of country-scale GHG emission estimates will remain difficult due to the lack of independent methods.

Mitigation options for grasslands can best be implemented at the farm scale. There is therefore a strong need for methods and models that allow precise estimations of GHG budgets at this spatial level and then to design and verify farm scale mitigation options. Currently, national inventories use a top-down approach, in which information about agricultural practices is aggregated and then weighted by emission factors. Such methods are input-based, and therefore only reflect changes in inputs but are not sensible to management changes. In particular, efficiency improvement do not affect the national inventory, if they are not accounted for by changes in emission factors. Thus, a bottom-up approach, i.e. farm approach, could function as an incentive for the stakeholder. In addition, since mitigation options have to be taken by farmers, a farm-scale model might give the required insight into the trade-offs between the different GHGs. Finally, a whole farm approach allows to take into account the heterogeneity of farming practices.

GHG fluxes and mitigation options for grasslands are examined, considering CO_2, N_2O and CH_4 in turn and then in combination. Concerning CO_2 fluxes, first measurements of net ecosystem exchange indicate a sink activity for carbon at most grassland sites. However, the main part of the corresponding carbon storage occurs in short-lived products (such as hay and silage) which are harvested from the grassland plots. The kinetics of carbon accumulation following change in land use or in grassland management are: a) non-linear: they are more rapid during the early years after adopting a practice which enhances accumulation. b) asymmetric: for example, the accumulation of organic carbon after sowing a grassland is slower than the release induced by conversion from grassland to arable. These characteristics have several practical consequences:

I. Any estimate of C storage must refer both to the previous management and to the current management.

II. Rates of C sequestration expressed in t C ha^{-1} yr^{-1}, are highly dependent upon the duration to which they apply;

III. At equilibrium, accumulation no longer increases, but stock conservation requires maintenance of the practices which enabled its accumulation;

IV. The cessation or temporary interruption of stock-enhancing practices usually results in a rapid release.

Climate change is likely to interfere with the conservation of existing soil carbon stocks and with mitigation strategies aiming at storing organic carbon in soils. However, the negative effects of climate change on C stocks due to higher temperatures may be counterbalanced by increases in grassland productivity resulting both from the rise in atmospheric CO_2 concentration and from technology factors. In the future, without incentives for carbon sequestration (e.g. under Article 3.4 of the Kyoto Protocol), soil organic carbon content in European grasslands is likely to decline as a result of a reduced grassland area and of an intensive agricultural use of the remaining grasslands.

For reducing emissions of nitrous oxide there are a number of options that offer significant GHG mitigation, most of which rely upon better fertiliser (mineral and organic) use and water management. Reducing N surpluses both at the plot and at the farm scale allows to reduce N losses to the environment and thereby mitigate GHG emissions from soils and from farm buildings. For methane, the livestock and manure management sectors offer the greatest mitigation potential, with a need to consider animal breeding and diets, as well as biogas production from farmyard manure.

While there are clear prospects for mitigation of the individual GHGs from grassland, there are clear trade-offs and synergies between different greenhouse gases, which are only now beginning to be quantified. Increased soil carbon storage associated with increased fertiliser use can be offset by increased nitrous oxide emissions, while changes to reduce CH_4 emissions may cause similar interactions. It is therefore important to assess potential mitigation options for their impact upon all greenhouse gases.

Increasing our knowledge of GHG emissions can not go without a better understanding of the phenomenon at the process level, in particular those dealing with grassland soil functioning and, at the same time develop holistic approaches to study carbon and nitrogen cycles simultaneously. There are also strong needs in developing tools and methods for data inventory, collation and aggregation in order to summarise multi-site experiments results into more precise and flexible emission factors. Mitigation options should be further developed with an emphasis on the socio-economic aspects if we aim to propose scenarios of GHG mitigation that are compatible with the needs of the agricultural sector. Finally, potential climate change effects on grassland GHG emissions have to be explicitly taken into account, as well as the role of plant and soil biota diversity which mediates some of the interactions between climate change and GHG emissions in grasslands.

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• Abbreviations & Symbols

BNF	Biological N Fixation
C	Carbon
CAP	Common Agricultural Policy
CH_4	Methane
CO_2	Carbon dioxide
C-eq	Carbon equivalent
CRU	Climat Research Unit
DEFRA	Department for Environment Food and Rural Affairs (UK)
DM	Dry Matter
EC	European Commission
EEA	European Envrionment Agency
EF	Emission Factor
EPA	Environmental Protection Agency (US)
ERA	European Research Area
EU	European Union
FAO	Food Agricultural Organisation
FYM	Farm Yard Manure
GHG	Greenhouse gases
GWP	Global Warming Potential
INRA	Institut National de la Recherche Agronomique
IPCC	International Panel on Climate Change
LSU	Livestock Unit
N	Nitrogen
N_2O	Nitrous oxide
NBP	Net Biome Productivity
NEP	Net Ecosystem Production
NEE	Net Ecosystem Exchange
NH_3	Ammonia
NO	Nitric oxide
OM	Organic Matter
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
UNFCCC	United Nations Framework Convention on Climate Change
UK	United Kingdom
US	United States of America
WTO	World Trade Organisation

• Introduction

1.1 Aims of the report

This report was initiated at a meeting of invited experts held in Clermont-Ferrand, France - 4th & 5th September 2003 - under the auspices of the CarboEurope-GHG programme. The meetings between the grasslands and croplands focus groups were held jointly, since many management decisions are taken at levels that affect both grasslands and croplands, but this report is for grasslands only. A separate report has been prepared for croplands. The aim of this report is to:

- Provide up-to-date scientific information on the extent of greenhouse gas fluxes from European grasslands, and the factors controlling GHG (Greenhouse Gas) emissions;
- Examine the ways in which GHG emissions from grasslands are currently estimated and suggest, when appropriate, possible improvements;
- Examine possibilities to mitigate GHG fluxes from European grasslands and livestock farms;
- Identify key uncertainties and areas for future research.

The report is aimed at scientists and policy-makers involved in estimating GHG emissions from agriculture and in assessing mitigation measures to reduce these emissions.

The main sources of data and input presented in this report come from presentations made by the invited experts in Clermont-Ferrand. They refer to scientific literature as well as recent results from several European projects funded by the European Commission, Directorate General Research under the 5th Framework Programme in the area "Energy, Environment and Sustainable Development" (Key Action on Global Change, Climate and Biodiversity):

- ATEAM (Advanced Terrestrial Ecosystem Analysis and Modelling) - project n° EVK2-2000-00075.
- CARBOMONT (Effects of Land-use Changes on Sources, Sinks and Fluxes of Carbon in European Mountain Areas) - project n° EVK2-CT2001-00125.
- GREENGRASS (Sources and Sinks of Greenhouse Gases from managed European Grasslands and Mitigation Strategies) - project n° EVK2-CT2001-00105.
- MIDAIR (Greenhouse gas mitigation for organic and conventional dairy production) - project n° EVK2-CT-2000-00096.

1.2. Grasslands in Europe: sinks or sources?

World-wide grasslands and rangelands cover about 3500 million ha, more than the double of arable land. On the European continent, by contrast, 37% of the total area managed by agricultural practices is devoted to grasslands (Bourgeois *et al*. 2002). However, the ratio of arable to grassland is highly variable among European countries: above 5 in Scandinavia and below 0.8 for Ireland, UK and Austria (Table 1). Half of the total area classified as grasslands is indicated as natural grasslands.

In the EU 15, there were in the year 2000 about 55 million ha of grasslands. Since 1990 (the reference year in the Kyoto protocol) some 3 million ha grassland (-5% within 10 years) were converted to arable land, especially for maize cultivation (Carlier et *al*. 2004). FAO estimates cattle stock in Europe to 100 millions in 2003, that is a 20% decline within 10 years (120 millions in 1991). Following the new membership of 10 countries and the anticipated membership of Bulgaria and Romania, the EU will have an enlarged area of grassland of about 20 million ha (+36%) (Carlier *et al.,* 2004).

Besides their natural aspect, grasslands have a pure agricultural destination as a primary food source for wild herbivores and domesticated ruminants. Actually, grasslands being a mixture of different grass species, legumes and herbs may act as carbon sinks, erosion preventives, bird directive areas, habitat for small animals, nitrogen fixation (Carlier et *al*. 2004).

As such, most grasslands tend to have a positive environmental role. However, intensively managed grasslands tend to release nitrate to the groundwater and are also (together with the associated livestock for which the grassland is maintained) a major source of ammonia [1](DEFRA, 2002).

Croplands (i.e. lands used for the production of arable crops) are estimated to be the largest biospheric source of carbon loss to the atmosphere in Europe each year. The EU-15 estimates for the CO_2 cropland emissions (~78 Mt C y^{-1}) are of the same order of magnitude as the reported emissions of N_2O from agricultural soil (~60 Mt C-eq. in 2000) and CH_4 from agriculture (~50 Mt C-eq. in 2000). Grasslands, by contrast, are estimated to store carbon, but the grassland estimate, which is derived from a simple model CESAR (Vleeshouwers & Verhagen, 2002), is the most uncertain (coefficient of variation of 130%) among all land-use types (Janssens *et al.*, 2003). It has been estimated that grasslands in Europe (as far east as the Urals) gain 101 Mt C y^{-1} (Janssens *et al.*, 2003). However, the size of the estimated amount of carbon stored by grassland in a business as usual scenario (0.52 tC ha^{-1} yr^{-1}) is similar to the average amount of carbon lost when converting tilled cropland to grassland (Soussana et *al.,* 2004b). Since this is an extreme land-use change, it suggests that current estimates of C sequestration by grasslands may be too high.

Moreover, grasslands contribute to the biosphere – atmosphere exchange of non CO_2 radiatively active trace gases, with fluxes intimately linked to management practices. Of the three greenhouse gases that are exchanged by grasslands, CO_2 is exchanged with the soil and vegetation, N_2O is emitted by soils and CH_4 is emitted by livestock at grazing and can be exchanged with the soil (Figure 1).

The magnitude of these greenhouse gas exchanges with the atmosphere may vary according to several factors: climate, soil, vegetation and management. One recent estimate of N_2O fluxes from grasslands indicates a mean emission of 2.0 kg N_2O-N ha^{-1} yr^{-1} in 2000, which translates into 0.25 t CO_2-C equivalent ha^{-1} yr^{-1} (Freibauer et *al*., 2004).

[1] *Ammonia emissions from cattle, sheep, goats… account for >50% of EU ammonia emissions, and grasslands are maintained for these livestock.*

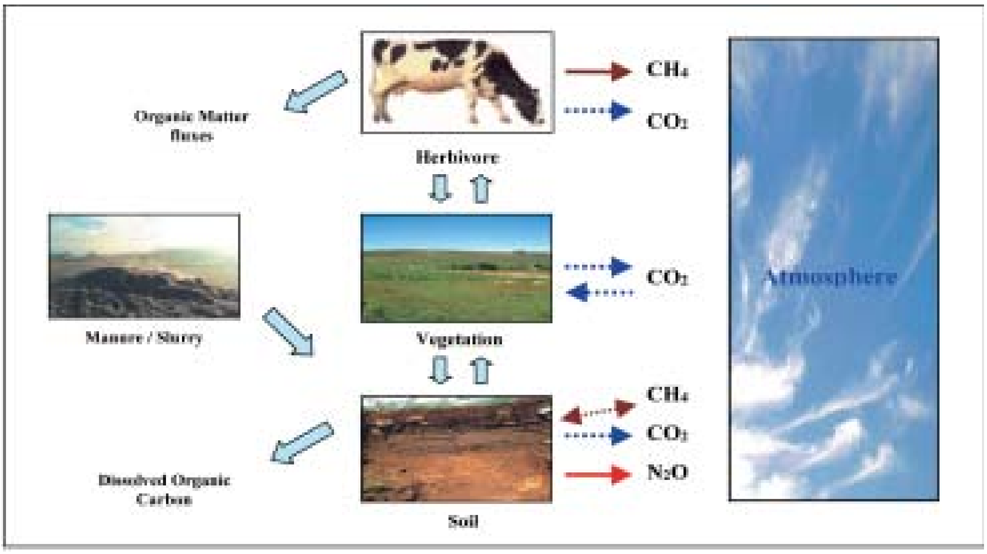


Figure 1 - *Greenhouse gas exchange and organic matter fluxes in grasslands*

Table 1 – *Managed grasslands and arable lands in the world and in EU (millions ha for 2000) after Bourgeois et al. (2002)*

	Arable	Managed grassland	Relation Arable/Grassland	Forests
World	**1512**	**3425**	**0.44**	**4080**
Europe	**310**	**181**	**1.71**	**155**
EU-15	85	56	1.54	106
Austria	1.5	1.9	0.79	
Belgium-Luxembourg	0.8	0.7	1.14	
Denmark	2.4	0.3	8.0	
Finland	2.2	0.1	22.0	
France	19.5	10.4	1.88	
Germany	12.1	5.3	2.28	
Greece	3.9	5.2	0.75	
Ireland	1.1	3.3	0.33	
Italy	11.1	4.3	2.58	
The Netherlands	0.9	1.0	0.82	
Portugal	2.6	1.2	2.17	
Spain	18.5	11.4	1.62	
Sweden	2.8	0.5	5.6	
United Kingdom	6.3	11.2	0.56	
C & E Europe	47	20	2.35	49
Russia	213	354	0.60	945
N & C America	**268**	**367**	**0.73**	**685**
USA	179	239	0.75	
Canada	46	29	1.59	
South America	**116**	**503**	**0.23**	**910**
Asia	**556**	**1055**	**0.53**	**540**
China	136	400	0.34	
India	170	11	15.4	
Oceania (Australia)	**202**	**888**	**0.23**	**650**
Africa	**54**	**418**	**0.13**	**195**

Grassland ecosystems are particularly complex and difficult to investigate because of the wide range of management and environmental conditions to which that they are exposed. Currently, the net global warming potential (in terms of CO_2 equivalent) from the greenhouse gas exchanges with grasslands is not known. From Figure 1, it is clear that an integrated approach, that would allow to quantify the fluxes from all three radiatively active trace gases (CO_2, CH_4, N_2O), would be desirable. However, current knowledge is scant. Management choices to reduce emissions involve important trade-offs: for example, preserving grasslands and adapting their management to improve carbon sequestration in the soil may actually increase N_2O and CH_4 emissions.

The Marrakech Accords, resulting from the 7th Conference of Parties (COP7) to the 1992 United Nations Framework Convention on Climate Change (UNFCCC), allow biospheric carbon sinks (and sources) to be included in attempts to meet Quantified Emission Limitation or Reduction Commitments (QELRCs) for the first commitment period (2008-2012) outlined in the Kyoto Protocol (available at: www.unfccc.de). Under article 3.4 the following activities are included: forest management, cropland management, grazing land management and re-vegetation. Soil carbon sinks (and sources) can therefore be included under these activities. Further, direct emission reductions of the greenhouse gases N_2O and CH_4 will help parties to meet QELRCs.

Parties electing to include grassland management, grazing land management and re-vegetation need to account for changes in these soil carbon sinks and sources on a net-net basis, that is to say, they must compare the net flux of carbon from a given activity during the commitment period with the equivalent net flux of carbon in the baseline year (usually 1990).

Carbon sequestration (*viz*. CO_2 fixation) in grassland soils, or even a reduction in a flux to the atmosphere compared to the baseline year, can therefore be used by a party to the UNFCCC in helping to meet emission reduction targets. Similarly, direct emission reductions of the greenhouse gases nitrous oxide (N_2O) and methane (CH_4) from croplands can also be used. It is essential that effects of land management of all three GHGs are evaluated concomitantly.

1.3. Greenhouse gas exchanges with grasslands: processes and fluxes

1.3.1. CO_2 and organic carbon fluxes

Carbon dioxide is lost from grassland soils by soil and root respiration and the decomposition of soil organic matter. Changes in organic carbon content are a function of the balance between inputs to soil of carbon fixed by photosynthesis and losses of soil carbon via decomposition. Rates of carbon input will therefore be dependent on the vegetation for both the managed grassland and native ecosystem. Soil erosion can also result in the loss (or gain) of carbon locally, but the net effect of erosion on carbon losses as CO_2 for large areas on a national scale is unclear.

For soils, both the quantity and quality of organic matter inputs and the rate of decomposition of soil organic carbon will be determined by the interaction of climate, soil and land use/management (including land-use history) (Figure 2). In native ecosystems, climate and soil conditions are the primary determinants of the carbon balance, because they control both production and decomposition rates.

In agricultural systems, land use and management act to modify both the input of organic matter via residue production, organic fertiliser application, grazing management and the rate of decomposition (by modifying microclimate and soil conditions through crop selection, soil tillage, mulching, fertiliser application, irrigation and liming) (IPCC, 1997). Management practices that increase soil and root respiration cause short-term effluxes of CO_2 to the atmosphere, whilst practices that increase the rate of decomposition of organic matter lead to longer-term losses of soil organic carbon in the form of carbon dioxide. Herbage harvesting by cutting also results in carbon exports from grassland plots. Most of the carbon harvested and stored in hay or silage will be released as CO_2 to the atmosphere shortly after harvest.

Changing pasture vegetation composition by seeding legumes is known to increase Net Ecosystem Production relative to pastures with no legume component. Grassland management often involves the sowing of specific forage species or the incidental introduction of species that become weeds, and this potentially can influence both the quantity and quality of carbon input to the soil, as well as microbial decomposition and loss of carbon from the soil. Ogle et *al.* (2004c) evaluated the impact of weedy annual brome species (*Bromus japonicus* and *B. tectorum*) in grasslands of western north America identifying critical functional differences between the annual brome species and perennial grasses that were relevant to soil organic matter dynamics, including litter quality, phenology and growth characteristics such as timing of maximum production. This information was used to simulate soil organic matter dynamics and associated processes, and the results showed that these species had the potential to increase carbon storage under current climatic conditions due to greater lignin content of bromes and heightened early season growth, relative to perennial grasses. Determining the impact of specific forage species or weeds will be dependent on the functional characteristics of those species as demonstrated by Ogle et *al.* (2004c).

The annual net ecosystem production (NEP) of a temperate grassland is between 1 and 6 t C ha^{-1} yr^{-1} according to the radiation, temperature and water regimes, as well as to the nutrient status and to the age of the sward (IPCC, 1996c). Nutrient and water supply may limit the potential NEP. For grasslands, the nature, frequency and intensity of disturbance plays a key role in the C balance. In a cutting regime, a large part of the primary production is exported from the plot as hay or silage, but part of these C exports is compensated for by farm manure and slurry application. Under intensive grazing, up to 60 % of the above ground dry matter production is ingested by domestic herbivores (Lemaire & Chapman, 1996). However, this percentage can be much lower during extensive grazing.

The largest part of the ingested carbon is digestible (up to 75% for highly digestible forages) and, hence, is respired shortly after intake. Only a small fraction of the ingested carbon is accumulated in the body of domestic herbivores or is exported as milk. Large herbivores, such as cows, respire approximately one ton C per year (Vermorel, 1995). Additional carbon losses occur through methane emissions from the enteric fermentation.

The non-digestible carbon (25-40% of the intake according to the digestibility of the grazed herbage) is returned to the pasture in excreta (mainly as faeces). In most European husbandry systems, the herbage digestibility tends to be maximised by agricultural practices such as frequent grazing and use of highly digestible forage cultivars. Consequently, the primary factor which modifies the carbon flux returned to the soil by excreta is the grazing pressure which varies with the annual stocking rate (mean number of livestock units per unit area). Secondary effects of grazing on the carbon cycle of a pasture include: i) the role of excretal returns, concentrated in patches, for the SOM mineralisation and the N cycling, especially in nutrient-poor grasslands, ii) the role of defoliation by animals and of treading both of which reduce the leaf area and the canopy photosynthesis.

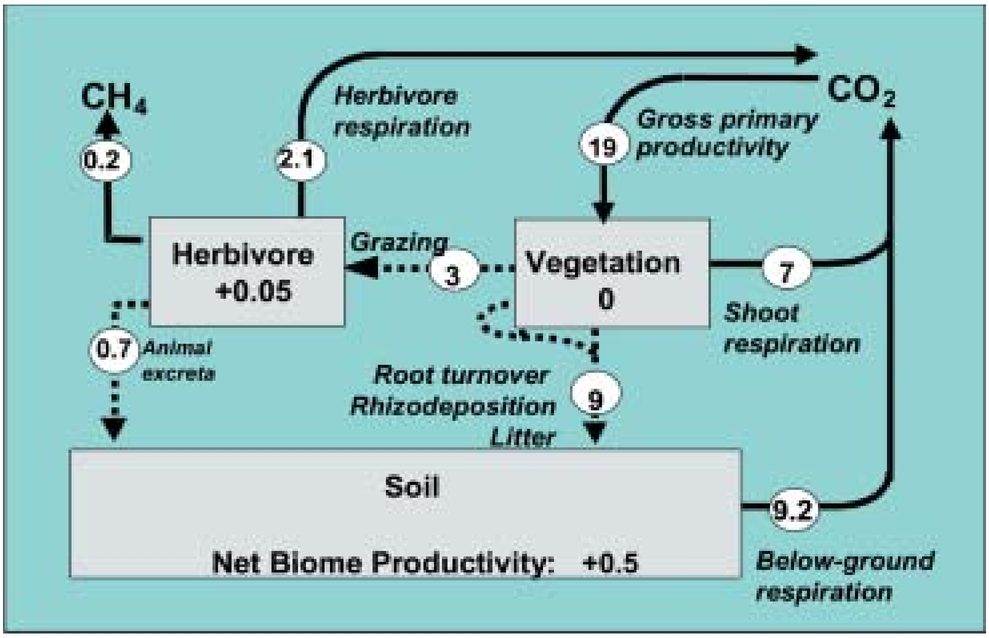


Figure 2 - *Carbon cycling in grazed grassland. The main carbon fluxes (t C ha^{-1} yr^{-1}) are illustrated for intensive grassland grazed continuously by cattle at an annual stocking rate of two livestock units per ha (Soussana et al., 2004b)*

Soil carbon stocks display a high spatial variability (coefficient of variation of 50%, Cannell *et al*., 1999) in grassland as compared to arable land and *ca*. 15% of this variability comes from sampling to different depths (Robles & Burke, 1998; Chevallier *et al*., 2000; Bird *et al*., 2002). Cropland soils are mixed through the action of the plow which also tends to reduce spatial variability relative to grassland soils. According Conant *et al.* (2001), in a recent review of soil carbon changes below temperate and tropical grasslands, a major factor accounting for changes in SOM content is the climate, because it affects differently the net primary productivity and the soil N mineralisation. This is also confirmed by research conducted by Amézquita *et al.* (2004) on soil carbon stocks on a range of tropical pasture and silvo-pastoral systems in sub-ecosystems of the American Tropical Forest. Their data also show that the level of soil carbon stocks in grasslands systems relative to native forest varies with the site altitude, temperature and precipitation.

The initial soil carbon content also accounts for part of the variability by being negatively correlated to the carbon stock change (Conant *et al*., 2001). By contrast, the soil texture does not seem to explain the variability between the different values of soil carbon contents observed. This last point is unexpected since numerous studies have shown a strong positive relationship between the soil carbon stocks and the fraction of clay or of clay plus fine silt (0-20µm) (Parton *et al*., 1987). Moreover, Reeder *et al.* (1998) have observed greater carbon storage after conversion from arable to grassland in a sandy soil compared to a clay soil. Therefore, more knowledge may be needed concerning the role of texture in the response of soil-C.

In grassland there is generally more soil organic carbon than under cropland (Cole *et al.* 1993) as a result of several factors including lack of disturbance, greater return of plant residues, high root biomass, manure application and the return of dung during grazing. As with arable crops, grazing practices which increase grassland productivity have the potential to increase SOM and C sequestration (Conant *et al.* 2001).

However this higher SOM concentration also provides the potential for larger losses of CO_2 and N_2O. Many intensively managed grasslands in western Europe also receive large inputs of fertiliser nitrogen often supplemented by inputs of organic manures (Smith *et al.* 2001). It has become apparent that these large inputs can be associated with significant nutrient losses both in the form of gaseous pollutants and losses in drainage water (Jarvis *et al.* 1996).

Organic matter is partly incorporated in grassland soils through rhizodeposition. This process favours carbon storage (Balesdent & Balabane, 1996), because direct incorporation into the soil matrix allows a high degree of physical stabilisation of the soil organic matter. Hutsch *et al.* (2002) have shown that up to 20% of fixed C can be released into the soil during the vegetative period. However our understanding of the underlying mechanisms and their quantitative contribution to overall soil C balance remains poor. The main reason is the difficulty in isolating the individual components experimentally, particularly under field conditions.

We need to improve our estimates of rhizodeposition of C from grasslands and our understanding of the mechanisms involved. This is important since the chemical nature of the C material deposited via exudation, secretion, cell sloughing and root tissue senescence will differ in a way that is likely to influence its fate in the soil. For example, in laboratory incubations, the mineralization rate of glucose and root mucilage-C is greater than that of root tissue-C (Mary *et al.* 1993).

Furthermore, a greater understanding of the physiological and molecular mechanisms involved in rhizodeposition would help identify opportunities for manipulating C inputs to the soil via the root system. It is known for example that the cutting of above ground vegetation can increase C loss from the roots (Paterson & Sim, 2000) and this suggests that there are opportunities through management (such as cutting and grazing) to alter rhizodeposition. Further research is needed to quantify the impact on net GHG emissions. Root turnover creates the largest organic carbon input to grassland soils and favours soil C storage, since root litter contains lignin and polyphenols which tend to be recalcitrant to degradation. Moreover, the soil organic matter is richer in aromatic compounds below a grassland than under a cereal monoculture, which confers on it a greater ability to resist degradation (Gregorich *et al*., 2001). After grassland establishment, roots and their associated microflora (bacteria and fungi) tend to stabilise the soil aggregates (Jastrow, 1996).

Therefore three reasons explain a greater C sequestration in grasslands than in arable soils: i) the absence of soil cultivation enables the development of physically protected as particulate organic matter (POM); ii) a greater part of this POM is chemically stabilised and iii) aggregates tend to protect the native soil organic matter from decomposition (Balesdent *et al*., 2000).

1.3.2. N_2O fluxes

Biogenic emissions of N_2O from soils result primarily from the microbial processes nitrification and denitrification. N_2O is a by-product of nitrification and an intermediate during denitrification (Figure 3). Nitrification is the aerobic microbial oxidation of ammonium to nitrate and denitrification is the anaerobic microbial reduction of nitrate through nitrite, nitric oxide (NO) and N_2O to N_2. Nitrous oxide is a gaseous product that may be released from both processes to the soil atmosphere (IPCC, 1997).

Major environmental regulators of these processes are temperature, pH, soil moisture (i.e. oxygen availability) and carbon availability (Velthof, 1997). In most agricultural soils, biogenic formation of N_2O is enhanced by an increase in available mineral nitrogen, which in turn increases nitrification and denitrification rates. Hence, in general, addition of fertiliser N or manures and wastes containing inorganic or readily mineralisable N, will stimulate N_2O emission, as modified by soil conditions at the time of application. N_2O losses under anaerobic conditions are usually considered more important than nitrification-N_2O losses under aerobic conditions. Therefore no-tillage will perhaps decrease CO_2 losses, but, due to poorer aeration, may enhance N_2O losses due to denitrification.

A schematic representation of N_2O losses from agriculture is given in Figure 2. Whilst N_2O emissions have been estimated in both process-based and inventory studies, the outstanding problem is the uncertainty of these estimates. The uncertainty is high because N_2O as CO_2 in soils are produced biologically and emissions usually occur in «hot spots» associated with urine spots and particles of residues and fertiliser, despite the diffuse spreading of fertilisers and manure (EEA, 2003). Six *et al.* (2004) conducted a meta-analysis reviewing the effect of no-tillage relative to conventional tillage methods on the net flux of the 3 major GHGs, and found that N_2O fluxes did increase in the first 5 to 10 years following adoption of no-tillage, presumably due to greater water filled pore space.

At a global scale, soils account for 65% of N_2O emissions (IPCC, 1996b). For given soil and climate conditions, N_2O emissions are likely to scale with the nitrogen fertiliser inputs. Therefore, the current IPCC (1996a) methodology assumes a default emission factor (EF_1) of 1.25% (range 0.25 to 2.25%) for non tropical soils emitted as N_2O per unit nitrogen input N (0.0025 - 0.0225 kg N_2O-N/kg N input). The emission factor of N in grazed grassland was higher (0.031 kg N_2O-N/kg N input) than for a cut grassland supplied with mineral fertiliser (Skiba *et al*., 1996). However, the default IPCC emission factor for N deposited during grazing is of 1% (EF_4).

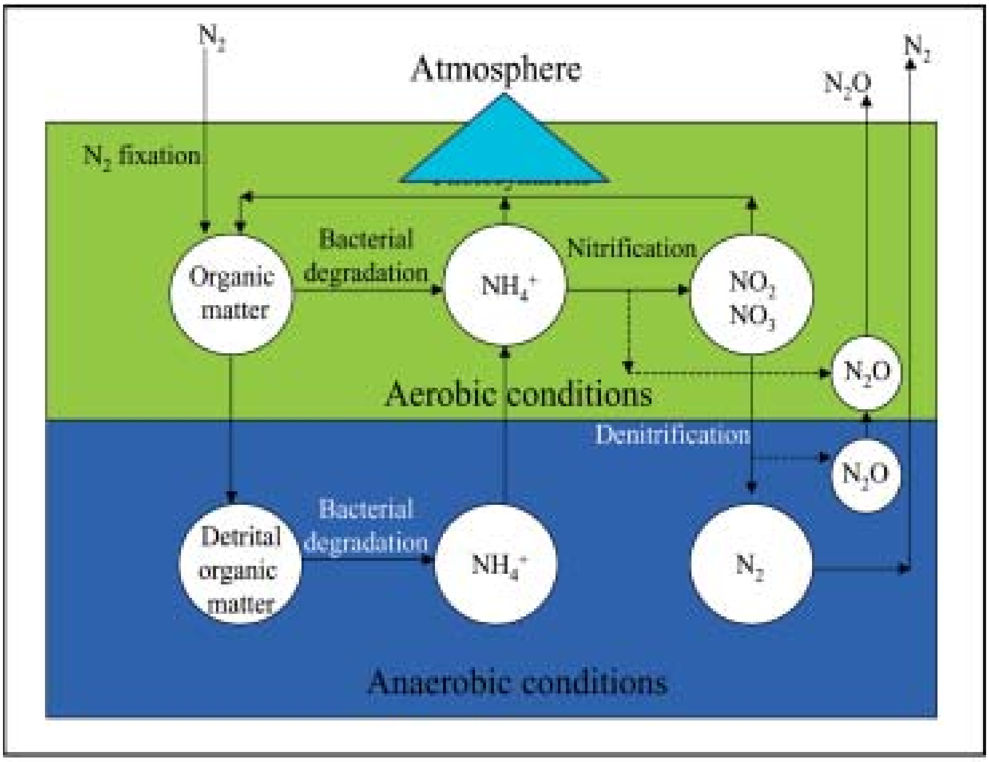


Figure 3 - *Microbial transformations in the soil showing mechanisms of N_2O production*

1.3.3. CH_4 fluxes exchanged with soils

In soils, methane is formed under anaerobic conditions at the end of the reduction chain when all other electron acceptors such as, for example nitrate and sulphate, have been used. Methane emissions from freely drained grassland soils are, therefore, negligible. In fact, aerobic grassland soils tend to oxidise methane, but less so than uncultivated soils with the oxidising capacity for forest, grassland and cropland soils showing the trend forests>grasslands>crops = 10 > 6 > 3 kg CH_4 ha^{-1} yr^{-1} respectively (Boeckx & Van Cleemput, 2001). For drained grasslands, methane oxidation was estimated between 0.1 and 1.1 kg CH_4 ha^{-1} yr^{-1} (Van Den Pol-Van Dasselaar, 1998).

1.3.4. CH_4 fluxes from enteric fermentation

The emissions of methane by ruminants are due to the fermentative reactions in the digestive tract, called enteric fermentation.

The rumen is a large anaerobic fermentative chamber located at the beginning of the digestive tract of ruminants, which contributes 70% of total organic matter digestion. Enzymes involved in ruminal digestion are solely of microbial origin. Fermentation of glucose equivalents from plant cell wall polymers or starch is an oxidative process under anaerobic conditions giving reduced co-factors such as NADH, which have to be re-oxidised to NAD^+ and H_2 to complete the fermentation of sugars (Figure 4). Although H_2 is one of the major end products of fermentation in pure cultures of rumen protozoa, fungi and bacteria, it does not accumulate in rumen gases. Hydrogen is immediately used by methanogens to produce methane in thermodynamically favourable reactions. For unknown reasons, acetogenesis is the main hydrogen sink in caecum and colon while methanogenesis is insignificant, thus explaining why the rumen plays a major role (95%) in total enteric methane emissions.

Every day a cow produces 300-700 litres of methane. The CH_4 emissions by cattle depend upon the type, age and weight of the animal and the quantity and quality of the feed consumed. Under grazing conditions, most of the variability in the enteric methane production of grassland plots lies in the number of animals, and therefore, the emissions per unit land area will primarily vary with the stocking rate.

Methane emission is positively correlated to the amount of fermented OM in the rumen and the intake of digestible energy (Blaxter & Clappperton, 1965). It increases with the amount of feed intake but, for the same diet, the proportion of gross energy lost in methane decreases with intake level. Large variations exist in methanogenic power of ingredients and diets. Methane production is closely related to the amount of dietary digestible cellulose content (Pinarès-Patino *et al*., 2003), whereas it decreases with addition of concentrate at a level higher than 30% in the diet (Giger *et al* 2000). Studies carried out on several animals fed on a same diet evidenced that as much as 40% differences in methane production appear between individuals. Reasons for such animal disparity remain unknown.

Impact of belched CO_2 (digestive + metabolic CO_2) on warming effect has not been considered until now as significant. Taking into account the GWP of each gas and their quantitative emissions, the warming CO_2/CH_4 ratio (WR) for various animal productions has been calculated. The WR varied from 1.1 for low-producing dairy cows to 1.8 for dry animals. Such original results indicate that CO_2 in belched gases affect more the global warming than methane.

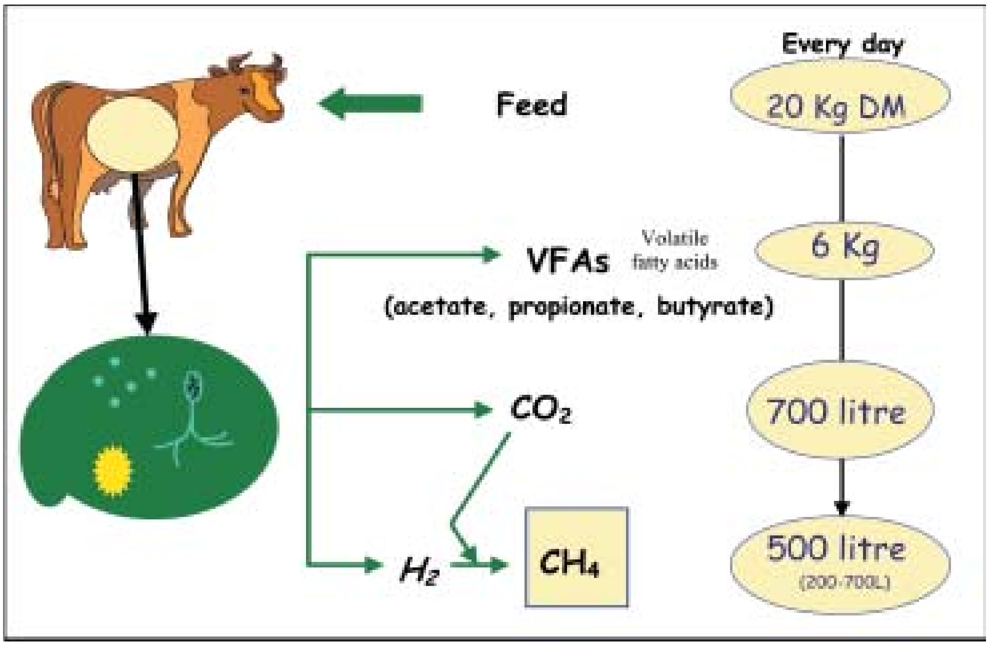


Figure 4 - *End-products of rumen fermentation*

CO_2 emission from cattle has not been taken into account by the IPCC because it is "short-cycling" carbon, which has been fixed by the plants earlier and has thus no effect on atmospheric concentrations. It should be noted that when animal feed is imported this leads to transboundary fluxes of carbon and to CO_2 emissions in the importing country. However, because it has been fixed by plants in the other exporting country these CO_2 emissions should not be counted in the importing country.

Ruminants are the major methane producers since they account for 95 % of the total enteric methane emissions. The emissions of methane by ruminants contribute between 16 and 23% of the global emissions of this gas (IPCC, 1996b). The annual emissions of CH_4 originating from enteric fermentation are typically between 80 and 100 kg animal^{-1} yr^{-1} for dairy cattle in Europe (IPCC, 1996a), leading to annual emissions equivalent to 0.67–0.84 t C per animal as CO_2 equivalent.

Enteric emissions in the world and the EU-15 have been estimated to 60-80 Tg and 7-10 Tg per year respectively, and contribute around 18 and 29 % of total methane emissions respectively. The enteric methane plays a greater part to total methane emissions in EU than in the world (29 *vs.* 18%). The direct contribution of enteric methane to the total greenhouse effect has been estimated to 2-3%. Enteric methane emissions tended to stabilise during the last decade at the world level, or even decreased in EU-15. Such a development within the EU is explained by a reduction of the number of animals and intensification of animal production following the reform of the European CAP.

1.3.5. Emissions during housing: manures and livestock wastes

On-farm emissions from animals and manure must be taken into account when the GHG mitigation potential of grassland management strategies involving grazing are evaluated. GHG emissions from manure management include direct emissions of CH_4 and N_2O, as well as indirect emissions of N_2O derived from NH_3/NO_x. Quantification of GHG emissions from manure are typically based on national statistics for manure production and housing systems combined with emission factors which have been defined by the IPCC or nationally (Petersen et *al.,* 2002). The quality of GHG inventories for manure management is critically dependent on the applicability of these emission factors.

Animal manure is collected as solid manure and urine, as liquid manure (slurry) or as deep litter, or it is deposited outside in drylots or on pastures. These manure categories represent very different potentials for GHG emissions, as also reflected in the methane conversion factors and nitrous oxide emission factors, respectively. However, even within each category the variations in manure composition and storage conditions can lead to highly variable emissions in practice. Figure 5 presents the effects of season, turning and dry matter content on GHG emissions for solid (a) and liquid (b) manure storage. This variability is a major source of error in the quantification of the GHG balance for a system. To the extent that such variability is influenced by management and/or local climatic conditions, it may be possible to improve the procedures for estimating CH_4 and N_2O emissions from manure (Sommer *et al*., 2004).

Excreta (dung and urine) deposited during grazing influences fluxes of both CH_4, N_2O and NH_3 from the pasture. In particular, urine patches are important point sources of NH_3 and N_2O, whereas the N input may locally reduce CH_4 oxidation activity. Ammonia losses from pastures are not specifically represented in the IPCC methodology, which calculates NH_3 volatilisation as a fixed proportion of total N excreted. However, ammonia losses from excretal returns to the pasture increase with N surplus in the diet since this N is mainly excreted as urea in the urine. Also, several methodologies exist for mitigating NH_3 losses from storage facilities. Hence, both optimised feeding and restricted access to grazing with collection of manure on the farm are available as NH_3 mitigation options, though not identified by the IPCC methodology. Technical solutions to reduce NH_3 volatilisation from storages may reduce (slurry) or increase (solid manure) CH_4 emissions (section 4.2.1), an aspect that must also be taken into account.

The N_2O emission factor for N deposited on pastures is higher than for N in manure collected during housing, indicating that restricted access to grazing is also a N_2O mitigation option. Several studies have suggested that N_2O emissions from excreta deposited during grazing interact with factors like feed composition, stock density, N fertilisation, soil compaction and climate. However, there is presently little evidence to suggest that emissions of N_2O can be consistently changed via management practices.

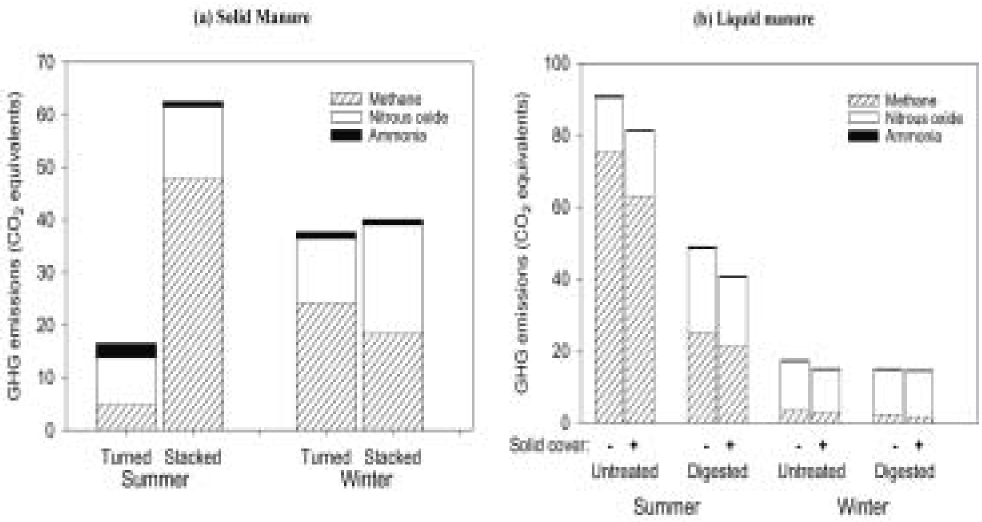


Figure 5 - *Effects of season, turning and DM content on GHG emissions for solid (a) and liquid (b) manure storage. After Ammon et al. (2003)*

1.3.6. Global Warming Potential (GWP).

When assessing the impact of land use and land use change on greenhouse gas emissions, it is important to consider the impacts on all greenhouse gases (Robertson *et al*., 2000; Smith et *al.,* 2001). Further, while animal production is not covered by this report, it should be emphasised that changes in manure management, such as the proportion deposited during grazing, may also influence the GHG balance of land use strategies. In order to assess the GHGs together, N_2O and CH_4 emissions are often expressed in terms of CO_2 or CO_2-carbon equivalents, which is possible because the radiative forcing of nitrous oxide, methane and carbon dioxide, can be integrated over different timescales and compared to that for CO_2.

This measure is called the Global Warming Potential (GWP). For example, over the 100-year timescale, one unit of nitrous oxide has the same global warming potential as 310 units of carbon dioxide, whereas, on a kilogram for kilogram basis, one unit of methane has the same GWP as 21 units of carbon dioxide (IPCC, 2001a). Currently, the net global warming potential (in terms of CO_2 equivalent) arising from the greenhouse gas exchanges with grassland is not known. An integrated approach is needed that would allow the fluxes of all three trace gases (CO_2, CH_4, N_2O) to be quantified (Figure 1).

• 2. Effects of agricultural practices on soil carbon stocks and GHG fluxes exchanged with grasslands

2.1. Effect of agricultural land use and management on soil carbon stocks

2.1.1. Global perspective

Soil organic carbon (SOC) storage in grasslands is impacted by climate, soil characteristics, topography, vegetation and management, but arguably management has had the largest impact on SOC storage in our modern world due to technological advances and population growth that has led to intensified management of agricultural lands. The most recent IPCC report dealing with this issue (IPCC, 2004) provides estimates for the grassland management on SOC storage based on a literature review and meta-analysis of grassland studies (Figure 6). According to those findings, land use change from grassland to cropland systems causes losses of SOC in temperate regions ranging from 18% (±4) in dry climates and to 29% (±4) in moist climates. Converting cropland back to grassland uses for 20 years was found to restore 18% (±7) of the native carbon stocks in moist climates (relative to the 29% loss due to long-term cultivation) and 7% (±5) of native stocks in temperate dry climates.

Based on the IPCC method for classifying management systems (IPCC, 2004), grassland practices are categorised as improved (e.g., sowing legumes, irrigation, fertilisation and planting more productive forage species) or degraded (e.g., overgrazing and planting less productive species relative to native vegetation), and two input classes are recognised for improved systems - medium input for grasslands managed with a single improvement and high input for grasslands managed with two or more improvements. Grasslands that are degraded for 20 years typically have 5% (±6) less carbon than native systems in tropical regions and 3% (±5) less carbon in temperate regions. Improving grasslands with a single practice caused a relatively large gain in SOC over 20 years, estimated as 14% (±6) in temperate regions and 17% (±5) in tropical regions, while having an additional improvement led to another 11% (±5) increase in SOC (see Ogle *et al*. 2004b for further discussion). However, based on the IPCC definition, a degraded grassland with less productive species, can have a high percentage of area covered with weeds - non-productive species for animal production purposes -, but showing high levels of soil carbon stocks. This was found by Amézquita *et al*. (2004) in their carbon sequestration research conducted in sub-ecosystems of the American Tropical Forest. This fact suggests that the reference system to account for soil carbon sequestration should be carefully defined.

The IPCC has provided a method in which these estimates can be used to estimate changes in SOC storage at the national scale (IPCC 1997, 2004), and Ogle *et al.* (2004b) found that by applying these factors in U.S. grasslands, SOC storage potentially could increase by 5 to 142 TgC yr^{-1} over 20 years. These values represent biophysical potentials, and the actual rate of increase will be dependent on current conditions of those grasslands and the willingness of managers to adopt conservation practices. Unfortunately, limited information is available regarding grassland conditions in many countries, and this is a major impediment for using management to offset emissions as part of international agreements. Assuming countries can track land use and management activity, adoption rates will be the second major determinant of how much carbon is sequestered in agricultural lands. Incentives are likely to play a major role in those rates through carbon trading markets or support programs offered by organisations interested in improving management such as governments or conservation groups.

In a review, Murty *et al.* (2002) showed that conversion of forest to cropland resulted in a loss of 22 ± 4.1% (n = 33) of the SOC. Conversion of forests to grasslands did not result in a significant SOC loss (SOC stock change is +6.4% ± 7.0%, n = 31). Guo & Gifford (2002) performed a meta-analysis study (537 observations from 74 publications) to assess the effect of land use change on SOC stocks. These authors indicated the following SOC stock changes: conversion of native forest to cropland (-42%), pasture to cropland (-59%), native forest to pasture (+8%), and cropland to pasture (+19%).

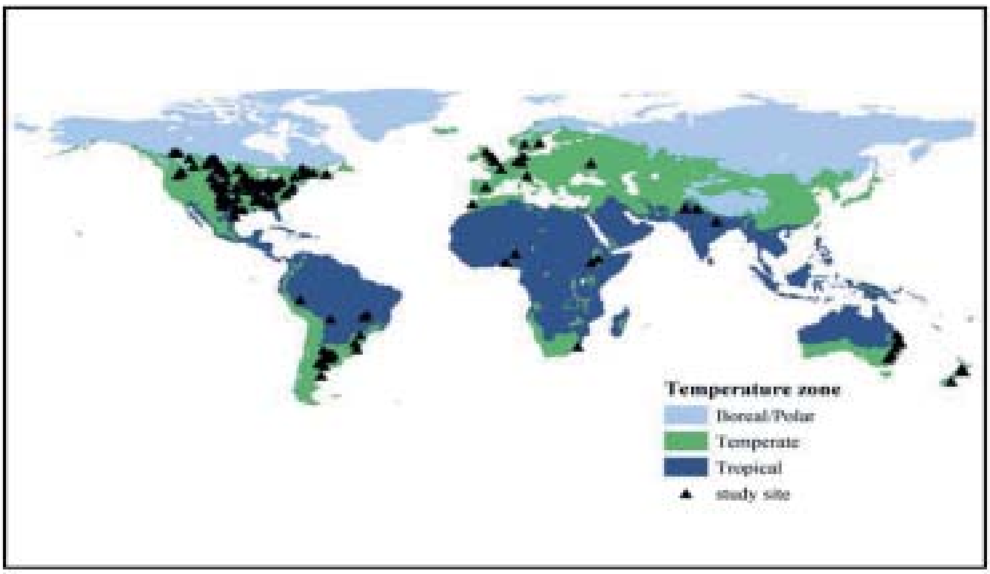


Figure 6 - *Agricultural Land Use and Management Studies*

For the conversion of pasture to cropland only the time after conversion and rainfall had an effect on SOC changes. For the conversion of cropland to pasture only sampling depth showed an effect. Murty *et al*. (2002) as well as Guo & Gifford (2002) indicated that the quantity of the available data is far from large enough. As a consequence only trends should be deduced from their analyses.

Vleeshouwer & Verhagen (2002), using a model study including the effects of crops (species, yield and rotation), climate and soil showed that by 2008-2012 and under *Business As Usual* C fluxes for agricultural areas in Europe are as follows: +52 g C m^{-2} yr^{-1} in grassland, -84 g C m^{-2} yr^{-1} in cropland and +144 g C m^{-2} yr^{-1} for cropland to grassland conversion. For the latter, Conant *et al.* (2001) presented a value of 101 g C m^{-2} yr^{-1}. Post & Kwon (2000) mentioned that reversion of cultivated land to grassland could result in a SOC accumulation rate of 33.2 g C m^{-2} yr^{-1}.

2.1.2. Effects of land use changes to or from grasslands on soil carbon stocks in Europe

In Europe, most soils are out of equilibrium as they have been affected by past land use / management practices. Management practices affecting GHG emissions from grassland areas include changes between arable and grassland, grassland and forest (and so on…), grassland management such as tillage (sown grasslands), grazing and cutting management, inorganic and organic fertiliser use, legumes, the type of fertiliser applied, water management…

Changes in soil carbon through time are non linear after a change in land use or in grassland management. A simple two parameters model can be used to assess such changes (INRA, 2002; Soussana *et al*., 2004b).

2.1.2.1. Modelling organic matter stock changes

The Hénin-Dupuis model (1945) which has a single carbon pool and two coefficients (one corresponding to the rate of conversion into humus of the OM added to the soil and the other to the rate of mineralisation of this humus) has been used to fit soil carbon stock changes calculated in a 0-30 cm depth. Carbon storage by converting management A to management B is determined according to two parameters:

- Δ, which is the stock difference at equilibrium, C_{eqB}-C_{eqA}
- *k*, a relative rate constant for carbon storage (in per year units)

The mean annual carbon storage flux (*F*) can be calculated for a duration *T* (in years):

$$F = \Delta\ [1-exp(-kT)]/T$$

Compared to a linear approximation, this exponential model has the following advantages:

I. it is closer to the kinetics which are actually measured,
II. it does not create risks of overestimating the carbon storage fluxes by extrapolII. lating the duration of the short term fluxes for too long,
III. the asymmetry between two land use changes can easily be quantified with this model.

2.1.2.2. Modelling soil carbon stock changes in a grass-crop rotation

The same simple statistical model can be applied with the following parameters :

I. C_0, initial carbon at t_0, the start of the simulation,
II. *Cc* (in t C ha^{-1}), equilibrium soil carbon stock under an annual crop monoculture,
III. *Cg* (in t C ha^{-1}) equilibrium soil carbon stock under a grassland monoculture (*Cg*>*Cc*).

The net accumulation rate (*C(t)*) of 'grassland' carbon after sowing a grassland (assuming a time constant *ks*) can be calculated from:

$$C(t) = C_0 + (Cg-C_0)\ (1-exp^{-ks\ t}) \qquad (1)$$

The net decomposition rate of carbon after tilling an existing grassland and sowing an annual crop (assuming an exponential time constant *kc*) can be calculated, assuming an initial carbon stock (C_1>*Cc*) at the end of the grassland phase ($t = t_1$):

$$C(t) = C_1 - (Cc-C_1)\ exp^{-kc(t-t1)} \qquad (2)$$

Since there is no simple analytical solution to these equations when *kc* differs from *ks*, numerical calculations of the soil organic carbon dynamics have been performed with a one year time step.

2.1.2.3. Grassland vs. arable land

The conversion of grasslands to arable has led to a 25-43% decline in soil carbon stocks in the uppermost 120 cm in the USA, as compared to the native grassland (Potter *et al.*, 2000). A well documented chronosequence in France has yielded similar results (Boiffin & Fleury, 1974). The mean carbon loss induced each year by converting a permanent grassland to an annual crop can reach –0.95 ± 0.3 t C ha^{-1} yr^{-1} over a 20-year period (Table 2 and Figure 7).

A three year period of bare fallow induced mean soil carbon losses of 1.7, 2.8 and 3.2 t C ha^{-1} yr^{-1} following an annual crop, a sown grassland and a permanent grassland, respectively, (Loiseau *et al.*, 1996). Hence, carbon losses tend to increase with the duration of the previous grassland phase.

The conversion of arable land to grassland resulted, according to IPCC (2000a), in a 0.5 t C ha^{-1} yr^{-1} average carbon storage over 50 years, with a range of 0.3 to 0.8 t C ha^{-1} yr^{-1}. Another meta-analysis (INRA, 2002) shows that, on average, for a 0-30 cm soil depth, carbon storage reached 0.49 ± 0.26 t C ha^{-1} yr^{-1} over 20 years (Table 4).

This rate of increase of soil carbon after conversion to grassland is slow. After 50 years, the soil carbon content is not restored to the level it had reached before grass was established (Rasmussen *et al*., 1998; Burke *et al*., 1995). Because of this slow accumulation, Franck (2002) considers that grasslands of more than 20 years no longer act as carbon sinks. The average time constant of carbon storage (0.025 ± 0.1 yr^{-1}), according to the fitted model, is less than half that of the carbon release rate after ploughing (Table 2).

Indeed, after 6 years of cultivation, Reeder *et al.* (1998) observed that soil carbon stocks had already reached the low values found after 60 years cultivation. Hence carbon losses are much faster after returning a grassland to arable than the build-up of soil carbon when establishing a grassland.

The increase in soil carbon content after a shift from arable to grassland is partly explained by a greater supply of carbon to the soil under grass, mainly from the roots but also from shoot litter, and partly by the increased residence time of carbon due to the absence of tillage. Carbon losses after tillage reduce the degree of physical protection of the organic matter, resulting in a decrease of the humified soil organic matter fraction (Post & Kwon, 2000). An increase of the soil disturbance caused by tillage increases the turnover of aggregates and accelerates the decomposition of soil organic matter within aggregates (Paustian *et al.*, 2000). After the establishment of grassland on an arable soil, a continuous vegetation cover and, hence, continuous protection of the soil organic matter contributes to an increase the soil carbon storage.

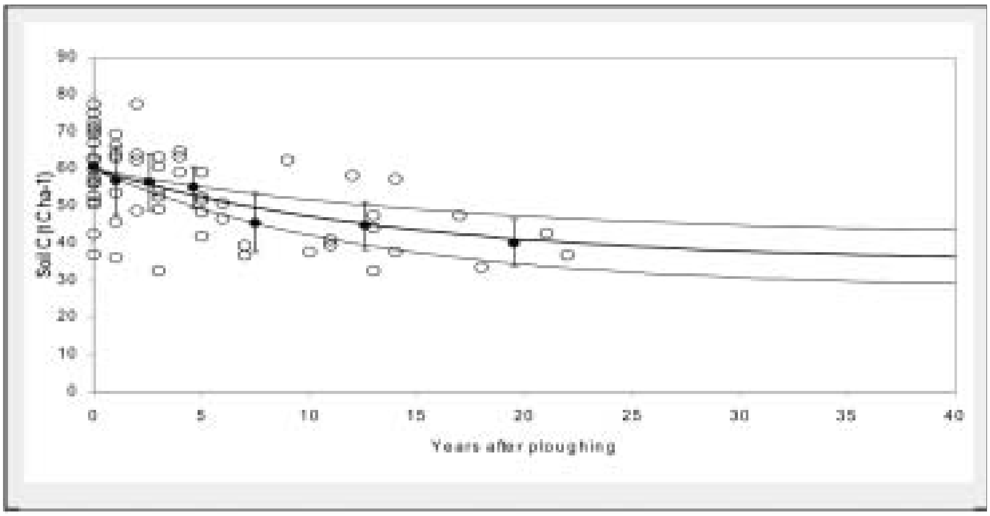


Figure 7 – *Decline in soil organic C content after converting a grassland to arable use. The data are from a chronosequence in Northern France. After INRA (2002).*

Table 2 - *Fitted values of the Hénin-Dupuis model for land use changes between arable and grassland. Carbon storage is determined according to two parameters the magnitude D (t C ha^{-1}), which is the stock difference at equilibrium CeqB-CeqA and a time constant (k). F20 is the average carbon storage flux (t C ha^{-1} yr^{-1}) during a 20 year period after the start of the land use change. Uncertainties are 95% confidence intervals on the regression slope.*

	Δ	k	Δ k	F20
Grassland to arable	-25±7	0.07-25±70.01	-1.7-25±70.8	-0.95-25±70.3
Arable to grassland	25±7	0.025-25±70.01	0.63-25±70.36	0.49-25±70.26

2.1.2.4. Grassland vs. forest

Converting grassland to forest can lead to an accumulation or to a release of soil carbon depending on the conditions (Post & Kwon, 2000). In some favourable conditions e.g. clay or calcareous soils in a mountain climate, an average accumulation between 0.1 and 0.2 tC ha^{-1} yr^{-1} over 30 cm has been reported by Moares *et al*. (2001) following afforestation of 200 year old grassland. However, this carbon storage can only be detected per unit mineral mass of the soil. In andosols, Ross *et al* (1999) have also measured strong accumulation rates of 0.2 t C ha^{-1} yr^{-1} over 25 years and 0.12 t C ha^{-1} yr^{-1} over 200 years.

However, under less favourable conditions i.e. warmer climate, sandy or acidic soils, a carbon loss has been measured after the conversion of grassland or moorland to forest (Aggangan *et al.*, 1998; Compton & Boone, 2000; Franzluebbers *et al.*, 2000). This literature survey leads us to propose small average carbon storage rates of 0.1 ± 0.02 t C ha^{-1} yr^{-1} over 20 cm during 90 years (INRA, 2002). There is very little information on the effects of deforestation followed by the establishment of pastures or grasslands for temperate zones. In New Zealand, the establishment of grasslands on old degraded forest soils has allowed an increase in the soil carbon stocks (Haynes & Williams, 1993). Because of a lack of appropriate information, carbon stock changes after converting forest to pasture may be considered to be symmetric to the inverse change, with a higher degree of uncertainty however: -0.1 ± 0.1 t C.ha^{-1} (INRA, 2002).

2.1.2.5. Short duration leys (sown grasslands)

As a result of periodic tillage and resowing, short duration grasslands tend to have a potential for soil carbon storage intermediate between crops and permanent grassland. Part of the additional carbon stored in the soil during the grassland phase is released when the grassland is ploughed up. The mean carbon storage increases in line with prolonging the lifespan of covers, *i.e*. less frequent ploughing.

Loiseau *et al*. (1996) studied carbon losses from sandy soils by comparing a permanent grassland, a crop-ley system (11 years and 9 years annual crop) and annual cropping systems. After 20 years, soil organic carbon stocks reached 24, 31 and 38 t C ha^{-1} for the arable, crop-ley and permanent grassland systems, respectively. Hence, introducing a ley into the rotation increased the soil carbon stock by 7 t C ha^{-1} after 20 years, which is approximately half the increase in soil carbon stock when arable is changed to permanent grassland (Loiseau *et al.*, 1996). Establishing a grassland for 3 years in a crop rotation leads to the additional storage of 3.5 t C ha^{-1} within 9 years (Lubet & Juste, 1979). This carbon storage potential is however strongly affected by the type of grassland management (see below).

2.1.2.6. Grassland management and ley-farming system effects on carbon stocks

To our knowledge, the effects of grassland management on carbon accumulation in soils have not yet been reviewed. To make some progress, based on an expert assessment for France (INRA, 2002; Soussana *et al.* 2004b), we try to give some of the possible trends in soil carbon caused by changes in grassland management. For France, different types of grasslands have been classified according to their mean organic carbon stock (INRA, 2002). Grassland types (Table 3) were defined according to the vegetation type (A to D) and to the nitrogen status of the vegetation (estimated by the nitrogen nutrition index method, Lemaire & Gastal, 1997). As sown grasslands usually have a higher nitrogen status than permanent grasslands, the most frequent types occur in the table along the diagonal. The following types have been described:

- A3- Short duration leys which are highly fertilised and used for cutting (>400 kg N ha^{-1} yr^{-1}) or grazing (>150 kg N ha^{-1} yr^{-1}) regimes. Losses of inorganic N are relatively large.

- A2- Short duration leys which are managed less intensively by cutting (<400 kg N ha^{-1} yr^{-1}) or grazing (<150 kg N ha^{-1} yr^{-1}) and which display a higher C/N ratio and, hence, less nitrogen losses than the A3 type.

- B2- Grass-legume mixtures or legume monocultures (e.g. lucerne) which are not supplied with nitrogen fertiliser. Under such conditions, nitrogen inputs to the soil tend to be self-regulated by ecological processes such as the inhibition of symbiotic nitrogen fixation, by soil inorganic N and the competitive decline of legumes under conditions of high soil N availability (Loiseau *et al.,* 2002).

- C2/D2- Grasslands, both sown or permanent, which are intensively managed for silage cuts or by intensive grazing (>1.5 LSU ha^{-1} yr^{-1})

- C1/D1- Species rich grasslands which have been extensively managed for hay production or for extensive grazing (<1.5 LSU.ha^{-1} yr^{-1}).

- D0- Nutrient poor grasslands and moorlands developed on organic soils at medium or high altitude. These grasslands are grazed with a low stocking rate (<0.8 LSU ha^{-1} yr^{-1}), without N fertiliser supply and are often invaded by shrubs or coniferous tree species.

Carbon stocks cannot be estimated in the grass ley system without taking into account the effects of the rotation between a crop and a sown grassland. Hence, mean soil carbon stocks were calculated at the end of the grassland and at the end of the crop stages for contrasting ley systems (Table 3). The time constants values in Table 4 have been estimated based on a previous assessment of humus types and soil organic matter inputs in contrasting grassland soils (INRA, 2002).

Table 3 - *A typology of the most common grassland types in France which has been developed to classify grasslands according to their mean soil carbon stock (INRA, 2002). This typology is based on the type of vegetation cover (A to D) and on the mean nitrogen status of the grassland vegetation. Nitrogen nutrition index values are between 0 and 1 and are calculated from the shoot nitrogen concentration and the standing biomass (Lemaire & Gastal, 1997).*

Type of grassland	Duration (years)	Nitrogen nutrition index			
		0.4 – 0.6	0.6 – 0.8	0.8 – 0.9	> 0.9
A. Short duration grass leys	1-2	-	-	A2	A3
B. Legume based leys	3-6	-	-	B2	-
C. Sown intensive grasslands	3-15	-	C1	C2	-
D. Permanent grasslands	>15	D0	D1	D2	-

[a] *These permanent grasslands are found at medium-high altitude*

Table 4 - *Simulated soil organic carbon stocks (t C ha^{-1}) at the end of the grassland phase, at the end of the annual crop phase and on average for contrasted ley farming systems. Ley farming systems were constructed by assuming a variable duration (left number, column 2) of the grassland phase with a given grassland type, followed by a variable duration (right number, column 2) of the annual crop phase. PG, permanent grassland. The time constant values were obtained by fitting the model to data of soil carbon stock changes (Soussana et al., 2004b). Simulations were run using the simple statistical model described in the Methods section. The uncertainty in these estimates is evaluated at ± 0.25 t C ha^{-1} yr^{-1}. Same abbreviations as in Table 3*

Grassland type	Ley system (yr)	Carbon stocks (t C ha^{-1})			Time constant (yr^{-1})	
		End of grassland	End of crop	Average	Start of grassland	End of grassland
A3	1/1	34.6	34.6	34.6	0.035	0.040
A2	2/1	44.3	44.9	44.6	0.035	0.040
B2	5/2	55.0	57.1	56.0	0.030	0.040
C2	5/2	53.4	55.3	54.3	0.025	0.040
C1	5/2	41.4	42.2	41.8	0.020	0.035
C2	10/2	59.0	61.2	60.1	0.025	0.040
C1	10/2	44.4	45.3	44.6	0.020	0.035
D2	PG	-	-	70	0.025	0.040
D1	PG	-	-	50	0.020	0.035
D0[a]	PG	-	-	110	0.010	0.035

[a] *These permanent grasslands are found at medium-high altitude.*

According to these estimates (Table 3 and 4), the grassland management strongly affects the soil carbon stocks. Moderate N fertiliser use increases the organic carbon input to the soil more than the soil C mineralisation. Intensive N fertiliser use induces not only a rise in production but also accelerates mineralisation and enhances decomposition of soil OM (Loiseau & Soussana, 1999) and, hence, reduces soil carbon stocks. Practices which enhance carbon stock are those which involve a reduction in the intensification of highly fertilised grasslands and a moderate intensification of poor grasslands.

However, mountain pastures and wetlands developed on organic soils should be excluded from the latter practice, because they are naturally endowed with high carbon levels that intensification could reduce by 1 t C ha^{-1} yr^{-1} (Loiseau *et al*., 1996). Under the conditions prevailing in Europe, grazing, which stimulates primary production, often increases carbon accumulation when compared with cutting (Table 4).

Some of the possible soil carbon sequestration opportunities for grasslands have been calculated and compared (Table 5) for 20 year time periods, by using the simple model with the time constants and carbon stocks values displayed in Table 6. According to these estimates, annual carbon storage rates between 0.2 and 0.5 t C ha^{-1} yr^{-1} are obtained for a range of options, which seem compatible with gradual changes in the forage production systems: reducing N fertiliser inputs in highly intensive grass leys, increasing the duration of grass leys, converting these leys to grass-legume mixtures or to permanent grasslands, or moderately intensifying nutrient-poor permanent grasslands. By contrast, the intensification of nutrient-poor grasslands developed on organic soils leads to large carbon losses and the conversion of permanent grasslands to leys of medium duration is also conducive to the release of soil carbon. Nevertheless, the uncertainties concerning the estimated values of carbon storage or release after a change in the grassland management are still very high (estimated at 0.25 t C ha^{-1} yr^{-1}) and further work will clearly be needed to ascertain the direction and magnitude of the soil carbon stock changes resulting from changes in grassland management. Moreover, it should be underlined that the values calculated here are mean values for French soils and cannot be extrapolated without caution to other temperate grassland areas which may display different soil and climate conditions.

Table 5 - *Soil carbon sequestration as affected by grassland management options. The effects on the soil carbon stocks after 20 years of changes in grassland management have been simulated with a one year time step using the values of soil carbon stocks, the time constants in Table 4 and the model described in the Methods section. Same abbreviations as in Table 3.*

	Initial type	Final type	Mean soil organic C stock after 20 yr (t C ha^{-1})	Soil organic carbon (t C ha^{-1})	C stock change (t C ha^{-1} yr^{-1})
Reduction of N fertiliser input	A3	A2	41.4	6.4	0.3
Conversion to grass-legume mixtures	A3	B2	45.6	10.2	0.5
	A2	B2	50.3	6.3	0.3
Intensification of permanent grassland	D1	D2	53.9	3.9	0.2
Intensification of nutrient poor grassland (organic soils)	D0	D2	87.4	-22.6	-1.1
	D0	C2 (10/2)	91.3	-18.7	-0.9
Permanent grassland to medium duration leys	D2	C2 (10/2)	67,0	-3,0	-0,2
Increasing the duration of leys	C2 (5/2)	C2 (10/2)	58.1	3.9	0.2
	C1 (5/2)	C2 (10/2)	50.9	9.1	0.5
Short duration leys to permanent grassland	C2 (5/2)	D2	60	5.7	0.3
	C1 (5/2)	D2	80	8.2	0.4

2.1.2.7. Case study: a comparison of arable and grassland systems effects on soil organic carbon

The focus of this study was on conversion of conventional tillage (CT) to reduced tillage (RT) in cropland and conversion of annual cropland systems to grassland or ley-farming systems.

A first case study has been conducted on a long-term field experiment (since 1966) (texture = sandy loam, no fertilisation) (Accoe *et al.,* 2002) consisting of permanent grassland (PG), continuous cropping (CC) and a ley-farming system (L) of three years annual crops (maize since 1981) followed by three years grass. The results showed that, with decreasing total SOC contents (in the order PG>L>CC; 22.5, 14.9 and 8.3 g C kg^{-1} in the 0-20 cm soil layer), the relative contribution of SOC associated with different size and density fractions decreased for all fractions except for the clay- and silt-sized fraction (<50 µm). This clearly reflected that SOC associated with the <50 µm fraction is physically better protected against degradation as a result of tillage.

In a second study (unpublished), the SOC content, distribution among size and density fractions and dynamics were compared in three fertilised grassland systems of different age (6, 14 and 50 years ago converted from cropland) (texture = loamy sand). The total SOC content (0-20 cm) was 17.1, 22.0 and 36.1 g C kg^{-1} soil for the 6-, 14- and 50-year old grassland respectively. Most of the SOC accrual was associated with the macro-organic matter fraction and the 50-150 µm size fraction. In the 14- and 50-year old grassland the $d^{13}C$ values of the macro-organic matter fractions were depleted relatively to the whole soil signal. Whereas the $d^{13}C$ values of the 50-150 and <50 µm fraction were slightly enriched compared to the whole soil signal. These trends indicate, that upon conversion of cropland to grasslands, freshly introduced SOC gradually shifts from the light to the heavy density macro-organic matter fractions and next into the 50-150 and <50 µm size fraction. However, the relative importance of the <50 µm fraction for SOC storage decreases with time, because the rate at which newly introduced SOC becomes protected by this fraction depends not only on the amount of free SOC introduced into the soil, but also on the degree to which the protective capacity of the clay- and silt sized fraction is filled up (enrichment ratio) (Hassink, 1996). The $d^{13}C$ signatures of the SOC fractions also indicated a decreasing turnover rate of SOC with increasing density in the macro-organic matter fractions and decreasing size fractions.

From review articles in the international literature it is clear that grassland soils could loose a significant portion of their SOC when converted to cropland. Alternatively, when cropland is converted to pasture the SOC content increased. However, in contradiction to Schimel *et al.* (2001), Guo & Gifford (2002) showed that SOC loss as a result of pasture to cropland conversion is larger than the SOC accrual during cropland to pasture conversion. These case studies demonstrated a significant SOC accrual after conversion of cropland to permanent grassland. Also ley-farming systems may enhance SOC sequestration relative to continuous cropping systems. However, here C sequestration could easily be offset by enhanced N_2O emissions. Soil organic C accrual was most significant for the macro-organic matter fraction and the 50-150 µm fraction. Changes in macro-organic material SOC can be explained by varying litter inputs in cropland compared to grassland systems. The importance of the 50-150 µm fraction in the SOC dynamics should be explained via the protective function of the formation of micro-aggregates. Thus, next to climate and land use change and management, physical protection of SOC importantly controls SOC sequestration. The latter is largely controlled by the amount of clay- and silt-sized particles (Hassink, 1996) and aggregate formation and turnover (Six *et al*., 2000).

The SOC content in permanent grasslands can be increased via specific management techniques. With respect to the latter we refer to Conant *et al.* (2001). In order to prevent SOC loss in cropland systems, attention should be paid to crop residue management (qualitatively and quantitatively) in combination with a reversion to reduced tillage cropland aiming at decreasing the micro-aggregate turnover rate and thus the potential for SOC build up (Six *et al.,* 2000).

2.2. Agricultural practices leading to non-CO_2 GHG emissions

The FAO states that climate, soil fertility (C content) and fertilisation are the most important drivers for N_2O emission from agricultural soils. In cropping management, both crop or forage type and soil wetness status are major influences on N_2O emission (Velthof *et al.*, 1997) and on CH_4 exchange (K. Smith *et al.*, 2000). These factors also influence the effect of a given cropland practice (Table 6 and 7) on non-CO_2 greenhouse gas emissions. For example, crop residues applied to a short-term grazed ley cannot be incorporated as they would be on land ploughed for arable cropping. Also minimum tillage of cereals into grassland rather than into previously ploughed land leaves a thatch of organic material near the surface. The different practices in each case result in a different mix and intensity of greenhouse gas fluxes. For these reasons, the emissions possible, even from well-researched practices, cover a wide range. One such practice is mineral fertiliser application where a typical soil N_2O emission from a ley is 2-10 kg/ha/year, but Dobbie & Smith (2003) reported up to 28 kg/ha/year at a site in Wales. Gas fluxes from less well-researched practices are even more uncertain, partly because of the paucity of data available and because the N and C composition of the manures and their evenness of spreading are so variable (Velthof *et al*., 1996). Up to 23 kg/ha N_2O have been measured after a single application of sewage sludge (Scott *et al.*, 2000).

Fluxes of GHGs from animal management, principally CH_4, are a little better understood, but are a function of a range of interacting factors, making it difficult to estimate flux from a given land area (Chadwick *et al.*, 2000) or even per animal. Thus, for each cropping practice, we have defined in Table 6 the likely rankings of emissions of N_2O, CH_4 and NO_x applicable to the land use practices presented in Table 7. For CH_4, the minus sign indicates atmospheric uptake and the plus sign indicates emission. NO_x (NO and NO_2) emissions are included as these gases increase tropospheric ozone production, thereby reducing the tropospheric CH_4 sink, and are precursors of acid rainfall. For each practice in Table 7 we have allocated a ranking for each gas. Note that in Table 7 a ranking for the importance of each practice is also given for carbon sequestration. This is low (x), medium (xx) or high (xxx). The practices listed in Table 7 refer to the addition of amendments (e.g. fertiliser), soil management (e.g. tillage), stock and crop management and inappropriate management (e.g. compaction). The table does not include indirect emissions of gases from drainage water and fresh water.

Table 6 - *Probability rankings of emissions of non-CO_2 greenhouse gases used in Table 2.2*

Ranking	Nitrous oxide (N_2O) (kg/ha/year)	Methane (CH_4) (kg/ha/year)	NOx, (kg/ha/year)
x Unlikely to exceed	5	± 1	1
xx Unlikely to exceed	10	± 2	2
xxx Could exceed	10	± 20	2

Table 7 - *Agricultural practices leading to non-CO_2 GHG emissions*

Practices	*Note: ranking is applied in rows only. It represents an average value for all items in each cell below*	N_2O	CH_4	NO_x
Inorganic fertiliser	• **Type and characteristics** - Ammonium providing fertiliser vs. nitrate fertiliser (mitigation technology) - Slow-release (mitigation tech.) - Inhibitors (mitig.)	xxx	x	xx
	• **Application techniques (mitigation)** - Synchronisation/timing (e.g. split-application) - Placement	xxx	0	xx
	• **Amount/rate of application** (recommandated rate…)	xxx	0	xx
Organic fertiliser	• **Type** - Farm yard manure, FYM (flux dependent on degree of maturity/degradation, moisture content) - Liquid fert./slurry (give higher fluxes than dried fert.) - Industrial waste - Household waste - Biogas residue (BR) - Fermented manure	xxx	xx	xx
	• **Timing** (less critical than inorganic)	xx	xx	x
	• **Storage processing and handling** - Temperature - Duration - Capacity, cover, etc. - Dimension - Aeration stage and crust	xxx	xx	xxx
	• **Amount/rate of application** (recommandated rate, crop dependent) - this is related to mitigation technology	xxx	x	xx
Biological N fixation	• **Mono cropping vs. Legumes / Grass mixture ratio** Most of the fluxes after ploughing; net effect is unknown	xx	0	xx
Crop residue	*(link with C sequestration)* • **Quality and size** (C :N ratio, total N; note that narrow ratio increases flux) • **Quantity** (weight) • **Application / incorporation techniques** (note the interaction with tillage; priming effect on soil N_2O flux mainly with incorporation)	xx	0	xx

Practices	*Note: ranking is applied in rows only. It represents an average value for all items in each cell below*	N_2O	CH_4	NO_x
Farming system & management	*(extensive/intensive - further research needed for the difference between extensive/intensive level; arable/livestock proportion)*	?	?	?
Tillage	*(link with C sequestration; interaction with soil physical condition)* • **Ploughing** (dependent on the available N at the time of ploughing) • **No tillage** (up to 10 kg N_2O-N/ha?); reduce NO (depend on moisture) • **Conservation** (reduced) tillage (intermediate?)	xx	x	xx
Crop rotation	*(mitigation technology)* • **Catch crops vs. bare soils** (possible link to BNF; increase N_2O) • **Amelioration crops** (crop type i.e. deep rooting or shallow rooting)			
Water management	*(drainage, irrigation, flooding water buffers, etc.)*	xxx	xx	xx
Compaction status	• **Inappropriate timing and over-sized machineries** • **Poaching**	xx	x	xx
Animal management	• **Animal management** • **Diet composition** • **Level of intake** • **Animal productivity/genetics** • **Intensity of system of animal production** • **Meat/milk production** • **Housing system** • **Storage capacity for manure** • **Age of slaughtering**	x	xxx	?
Grazing intensity	Including stock numbers and type of grass conservation	xxx	xxx	x
Local ammonia abatement		xx	0	x
Biomass burning & Biomass production		x	x	xxx

• 3. GHG balance at the plot scale

3.1. Annual GHG balance at the plot scale: first results from the GREENGRASS project

We present here some first results of the GREENGRASS' project (Soussana *et al*., 2004a), concerning the balance of CO_2, N_2O and CH_4 exchanges between grasslands and the atmosphere at 9 contrasted European sites that cover a major climatic gradient over Europe and including four grassland types (sown grass, sown grass-clover, intensive permanent grassland and semi-natural grassland), three types of herbage use (rotational cattle grazing, continuous cattle grazing and mowing) and contrasted N fertiliser supplies (Table 8).

Each consists of one (or more) grassland plot of several hectares. A bare soil (ploughed up permanent grassland) control and a barley crop are included for the purpose of inter comparison. Data from the Lelystad site are not shown here because of some missing values.

Table 8 - *Grassland type and management at the field sites*

Country		Name	Grassland type	Type of herbage use	Stocking rate/ cutting frequency	N fertiliser kg N ha^{-1}
Denmark		Lille Valby	1. Barley crop control	-	-	-
			2. Sown grass (in barley)	Cut	High	300
France		Laqueuille	Semi-natural grassland	Continuous grazing	1. High	190
					2. Low	0
Hungary		Bugac	Semi-natural dry grassland	Cut and grazed	Low	0
Ireland		Oak Park	Newly sown grass-clover	Cut and grazed	High	200
Italy		Malga Arpaco	Semi-natural grassland	Cut and rotationally grazed	Low	0
Scotland		Easter Bush	1. Intensive permanent grassland	Cut and rotationally grazed	High	200
		Poldean	2. Bare soil (ploughed up)	-	-	0
Switzerland		Oensingen	Newly sown grass	Cut	1. High	300
					2. Low	0
The Netherlands		Lelystad	Intensive permanent grassland	Cut and rotationally grazed	High	300

In each site, CO_2 fluxes are monitored close to the ground by open and closed path eddy covariance systems. N_2O fluxes are measured by GC-cuvette systems (weekly to fortnightly) and tunable diode laser (TDL) systems for eddy correlation measurements. CH_4 production from grazing cattle is measured *in vivo* in four of the grassland sites using the SF_6 tracer technique with 6-7 replicate animals. CH_4 exchanges with the soil are monitored with cuvette techniques. Grass and soil parameters are measured at regular intervals in order to characterise the sites, and to serve as model input data.

These include leaf area index, shoot biomass, *in vitro* digestibility and nitrogen content, soil heat flux, soil water content, soil carbon and nitrogen (organic and mineral forms). Continuous flux measurements for CO_2 and (semi-) continuous measurements for CH_4 and N_2O are made at the 9 sites since June 2002.

3.1.1. Net ecosystem exchange of CO_2

The net ecosystem exchange (NEE) of CO_2 was calculated from the continuous eddy flux measurements after performing quality analysis and quality check, gap filling and footprint analysis according to standardised procedures. A negative annual NEE denotes that the site is a sink for carbon during the year studied (Figure 8).

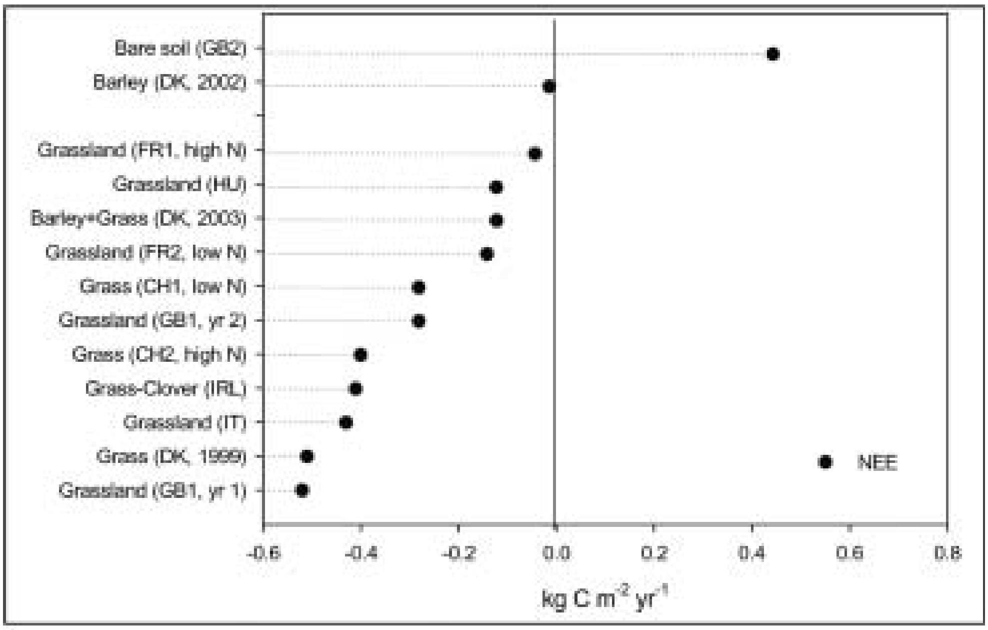


Figure 8 - *Sorted dotplot of the annual Net Ecosystem Exchange (June 2002 to June 2003) at the sites from the EC-FP5 GREENGRASS project.*

For the non-grassland covers, the NEE denotes a source (at the ploughed up site, GB2), or a very small sink (barley crop, DK). The transition from the barley to the grass in Denmark resulted in a slightly higher sink value.

3.1.2. N_2O and CH_4 emissions

The N_2O emissions measured in each plot were averaged over the year and converted into CO_2-C equivalents, assuming a global warming potential 300 times greater (per unit mass) than that of CO_2 over a 100 year horizon. Annual N_2O emissions varied from 0.004 to 0.24 kg CO_2-C equivalent m^{-2} yr^{-1} between the grassland sites (Figure 9), with emission factors (percentage of N fertiliser supply lost as N_2O) comprised between 0.5 and 2.0%.

Methane has a global warming potential which is 23 times that of CO_2 over a 100-year horizon. The methane emissions from the enteric fermentation of the cattle grazing the plots were calculated in CO_2-C equivalents per unit ground area and per year, from the average annual stocking rate and from the emissions per unit liveweight. Annual CH_4 emissions from cattle varied between 0.07 and 0.13 kg CO_2-C equivalent m^{-2} yr^{-1} (Figure 9). The net exchange of methane with the grassland soils was much smaller and is not shown here.

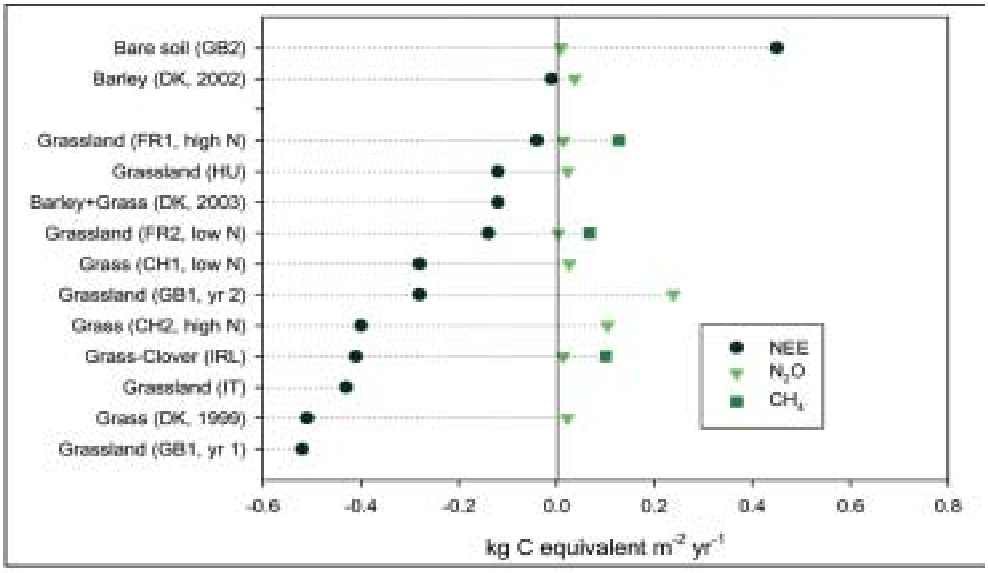


Figure 9 – *Annual NEE, N_2O and CH_4 emissions (in C equivalents) Preliminary results from the EC-FP5 GREENGRASS project*

3.1.3. Net balance of greenhouse gas exchanges

The net balance of GG fluxes exchanged with the atmosphere was calculated (when data were available) as the sum of the NEE and of the methane and nitrous oxide emissions. The results show that despite trade-offs between the net uptake of atmospheric carbon by the plots and the release of N_2O and CH_4, the GG balance of the grassland plots was indicating an overall sink activity (Figure 10), with the exception of the continuously grazed and intensively managed FR1 site.

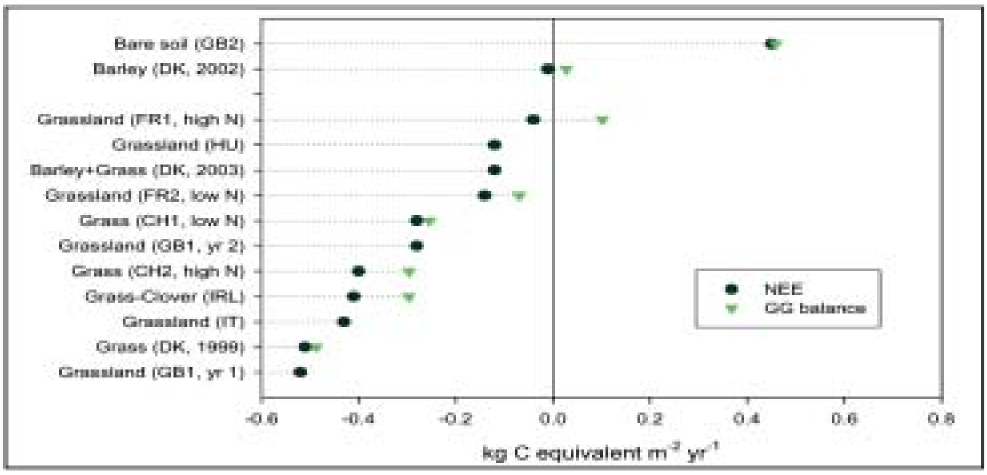


Figure 10 - *Net Ecosystem Exchange and GHG balance in C equivalents at sites of the EC-FP5 GREENGRASS project from June 2002 to June 2003. These preliminary results do not include the C release from short lived compartments such as hay and silage harvested from the grassland plots.*

3.1.4. Preliminary conclusions

For most sites, these results correspond to first year measurements. Subsequent years results may differ as GG exchanges are subjected to a large interactions with climate. Further checks will be performed, which may slightly alter the values displayed here. Despite these caveats, our first estimates clearly show that European grasslands may act as a sink for the exchange of radiatively active trace gases with the atmosphere.

The NEE from the grassland covers is in the same range as found for forest sites in Europe (Valentini *et al.,* 2000), with highest NEE values close to those found for coniferous (evergreen) forests. However, in both cases, the fate of the C products harvested (ie. wood in forests and forage in grasslands) has not been examined. With grasslands, the organic C harvested as hay or silage will be rapidly used for animal nutrition and returned to the atmosphere.

Janssens *et al.* (2003) concluded that European grasslands may constitute a net C sink (0.06 ± 0.08 kg C m^{-2} yr^{-1}), although the uncertainty surrounding this estimate was larger than the sink itself. Our findings for the grassland NEE show a significant ($p<0.01$) sink activity (0.30 kg C m^{-2} yr^{-1}) with a coefficient of variation of 57% among grassland sites.

The net biome productivity (NBP) of a grassland ecosystem is equal to NEE minus the loss of carbon by disturbance. Respiration of cattle grazing the grassland plots was included in the annual NEE estimates obtained by eddy covariance. NBP was thus close to NEE for these sites. By contrast, cutting and harvesting the herbage resulted in C exports which were not accounted for by the CO_2 flux measurements. Only a small fraction of the ingested carbon is accumulated in the body of domestic herbivores or is exported as milk.

The non-digestible carbon (25-40% of the intake according to the digestibility of the grazed herbage) is returned to the pasture in excreta. Assuming that 70% of the C harvested as hay or silage was respired within one year, the average annual NBP was estimated at -0.13 kg C m^{-2} yr^{-1} (range -0.40 to +0.11). An important conclusion from this approach is that there is no overall evidence yet from the flux measurements of a rapid increase in soil carbon at a rate comparable with the estimate by Janssens *et al.* (2003). However, it should be emphasised that these first results are only for one year of measurements and that the estimates will be further refined by the GREENGRASS project.

A further offset of the grassland sink activity occurred through the emissions of N_2O and CH_4. When converted in C equivalents, N_2O emissions reached on average, 16% (0.047 kg equivalent CO_2-C m^{-2} yr^{-1}) of the NEE. Moreover, at the three sites where estimates are available, the CH4 emissions from cattle reached 0.10 kg equivalent CO_2-C m^{-2} yr^{-1}, that is half the average NEE for these sites. This demonstrates that a full accounting of radiatively active trace gases is needed before being able to conclude about the role of grasslands for the greenhouse gas effect.

3.2. Mitigation options from plot and animal management

Mitigation options concerning non CO_2 GHG fluxes are summarised in Table 9. The role of changes in N fertiliser supply and in animal stocking density for the global warming potential, resulting from the net exchange of GHG with grazed grasslands is then discussed from modelling results. Strategies to identify and implement good agricultural practices that contribute to lower emissions of greenhouse gases from agriculture have been identified (Oenema *et al*., 2001; Oenema *et al*., 2004). Several research projects across Europe have identified options to mitigate greenhouse gases from agriculture through adopting specific good practices (Kuikman *et al*., 2003; Conant *et al*., 2001; Vellinga et *al*., 2004; Burczyk *et al*., 2001).

3.2.1. Mitigation of N_2O and CH_4 emissions

Table 9 - *Mitigation options for reducing non-CO_2 GHG fluxes*

Practices	*Note: ranking is applied in rows only. It represents an average value for all items in each cell below*	N_2O	CH_4	NO_x	Constraints
Inorganic fertiliser	• **Type and characteristics** - Ammonium providing fertiliser vs. nitrate fertiliser (mitigation technology) - Slow-release (mitigation tech.) - Inhibitors (mitig.)	xxx		xx	- Education - Availability & cost - Availability & cost
	• **Application techniques (mitigation)** - Synchronisation/timing (e.g. split-application) - Placement	xx			- Weather - Education - Labour & Capital
	• **Amount/rate of application** (recommended rate…)	xxx			- Education
Organic fertiliser	• **Type** - Farm yard manure, FYM (flux dependent on degree of maturity/degradation, moisture content) - Liquid fert./slurry (give higher fluxes than dried fert.) - Industrial waste - Household waste - Biogas residue (BR) - Fermented manure	xxx	xxx (for biogas)		- Education - Cost
	• **Timing** (less critical than inorganic)	xx	xx		- Education - Cost - Weather
	• **Storage processing and handling** - Temperature - Duration - Capacity, cover, etc. - Dimension - Aeration stage and crust	xx	xx	xx	- Education - Cost - Weather - Soil conditions
	• **Amount / rate of application** (recommandated rate, crop dependent) - this is related to mitigation technology	xx	xx	xx	- Weather - Education - Labour & Capital

Practices	*Note: ranking is applied in rows only. It represents an average value for all items in each cell below*	N_2O	CH_4	NO_x	Constraints
Biological N fixation	• **Mono cropping vs. Legumes / Grass mixture ratio** Most of the fluxes after ploughing; net effect is unknown)	xx?			- Cost
Crop residue	*(link with C sequestration)* • **Quality and size** (C :N ratio, total N; note that narrow ratio increases flux) • **Quantity** (weight) • **Application / incorporation techniques** (note the interaction with tillage; priming effect on soil N_2O flux mainly with incorporation)	xx			- Availability (link with animal production) - Cost - Education
Tillage	*(link with C sequestration; interaction with soil physical condition)* • **Ploughing** (dependent on the available N at the time of ploughing) • **No tillage** (up to 10 kg N_2O-N/ha?); reduce NO (depend on moisture) • **Conservation** (reduced) tillage (intermediate?)	-	-	-	- Need further research - Education
Crop rotation	*(mitigation technology)* • **Catch crops vs. bare soils** (possible link to BNF; increase N_2O) • **Amelioration crops** (crop type i.e. deep rooting or shallow rooting)	xx		x	- Cost - Need further research
Water management	*(drainage, irrigation, flooding water buffers, etc.)*	xxx	xxx		- Cost - Need further research
Compaction status	• **Inappropriate timing and over-sized machineries** • **Poaching**	x		x	- Cost - Education

Practices	*Note: ranking is applied in rows only. It represents an average value for all items in each cell below*	N_2O	CH_4	NO_x	Constraints
Animal management	• **Diet composition:** *increasing the dietary level of concentrate (but the proportion of concentrate is already high in EU!)* • **Level of intake:** *increasing voluntary feed intake of animals (genetic selection of animals and use of condensed diets)* • **Animal productivity/genetics:** - *Use of rumen by-pass ingredients (genetic selection and -or- technological treatments of plants)* - *Use of bovine somatrotopine (approved in the USA but banned in EU)* - *Genetic selection of low-methane animal producers (feasibility?)* - *Use of feed additives to stimulate feed efficiency (probiotics, antibiotics) or alter methane production (ionophore antibiotics, dicarboxylic acids, unsaturated fatty acids)* - *Use of some plant extracts (saponins, essential oils), immunisation of ruminants against methanogens or protozoa (Australian patent). It must be noted that the use of feed additives will be severely controlled in the future. As an example, all the antibiotics and chemical feed additives will be banned from year 2006 in all the EU countries.* • **Intensity of system of animal production** • **Meat/milk production:** *increasing the number of lactations per cow* • **Housing system** • **Storage capacity for manure** • **Age of slaughtering:** *slaughter beef cattle at a younger age (the weight of pre-weaning period without methane emission is higher)*	x	xxx		- Animal welfare - Cost - Education - Social demand
Grazing intensity	Including stock numbers and type of grass conservation: decreasing the livestock number (intensification to maintain the level of animal production or even decrease beef production for the benefit of pig or poultry production	x	xxx		- Policies

3.2.2. Mitigation options and GWP at grazing

A mechanistic grassland ecosystem model (Thornley, 1998; Riedo *et al*., 1998; Schmid *et al*., 2001) has been developed to model trace gas fluxes in relation to other important properties of grassland ecosystems. The model (PASIM) is fully dynamic and can be used to simulate above and below ground dry matter production of a perennial sward in relationship to fluxes of C, N, water, and energy. Five years of simulation were started from an equilibrium state. The grassland was assumed to be grazed continuously by cattle during the 5 months between DOY 141 and 292. Two scenarios were compared: 150kg N ha^{-1} yr^{-1} split in three applications (mid-May, mid-July, end August) and no fertiliser N (mitigation). Different mean annual stocking rates were also compared (from 0 to 2.4 livestock unit ha^{-1}, with a 0.4 step) (Soussana *et al.,* 2004b). However, the simulated stocking density was reduced whenever the daily cattle intake dropped below a threshold (13.5 kg DM LSU^{-1} day^{-1}) as a result of insufficient herbage available. Thus, the simulated mean annual stocking rate was found to be less than 2.4 LSU ha^{-1} (Figure 11).

The daily output data from the PASIM model included: net canopy photosynthesis, soil, plant and animal respiration, methane emission by cattle and N_2O emission from soil. The net annual emission of CO_2, N_2O and CH_4 and their balance were computed from these data in CO_2-C equivalent and expressed either per unit land area (Figure 11 A to D) or per livestock unit (Figure 11 E, F).

Irrespective of N input, the carbon balance (*i.e.* the net biome productivity) of the simulated grassland was negative during the first year of grazing, indicating a sink for atmospheric CO_2 (Figure 11 A, B). However, the magnitude of this sink declined with the mean annual stocking rate, while the CH_4 emissions increased proportionally to stocking rate. The N_2O emissions were greater with, than without N fertiliser, but were little affected by the stocking rate. For mean annual stocking rates less than 1.6 LSU ha^{-1}, the simulated net balance of the GHG emissions in CO_2 equivalents was always negative during the first year, denoting a sink activity.

After five years of grazing, the simulated GHG emissions displayed the same trends as during the first year of grazing in the N fertilised grassland (Figure 11 D). By contrast, the unfertilised grassland displayed a strong decline in herbage growth (data not shown) and a positive or nil carbon balance (Figure 11 C). The simulated net balance of GHG emissions in CO_2 equivalents was negative (sink) for the fertilised grassland below 1.2 LSU ha^{-1}, but was always positive or nil for the unfertilised grassland.

GHG fluxes were also calculated for the first year per unit livestock (Figure 11 E, F). The simulated methane emissions per livestock unit were approximately constant. By contrast, the magnitude of the CO_2 fluxes per LSU declined strongly as the stocking rate fell. Therefore, the GHG budget per unit livestock was close to zero at high stocking rates (Figure 11 E, F).

A sustained GHG sink activity after 5 years time was simulated for low stocking rates and N fertiliser inputs (Figure 11 D). The past land management also affected the greenhouse gas fluxes, as can be seen from the difference in emissions between year 1 (after several years of cutting) and year 5 (after 4 years grazing). Finally, when expressed per unit livestock the GHG balance appears to be small but usually positive (source) at high stocking rates, while a negative GHG balance (sink) is obtained with low stocking rates (Figures 11 E and F).

Such model predictions need to be validated with experimental data. Therefore, it is at present not possible to generalise from the conclusions of this modelling study.

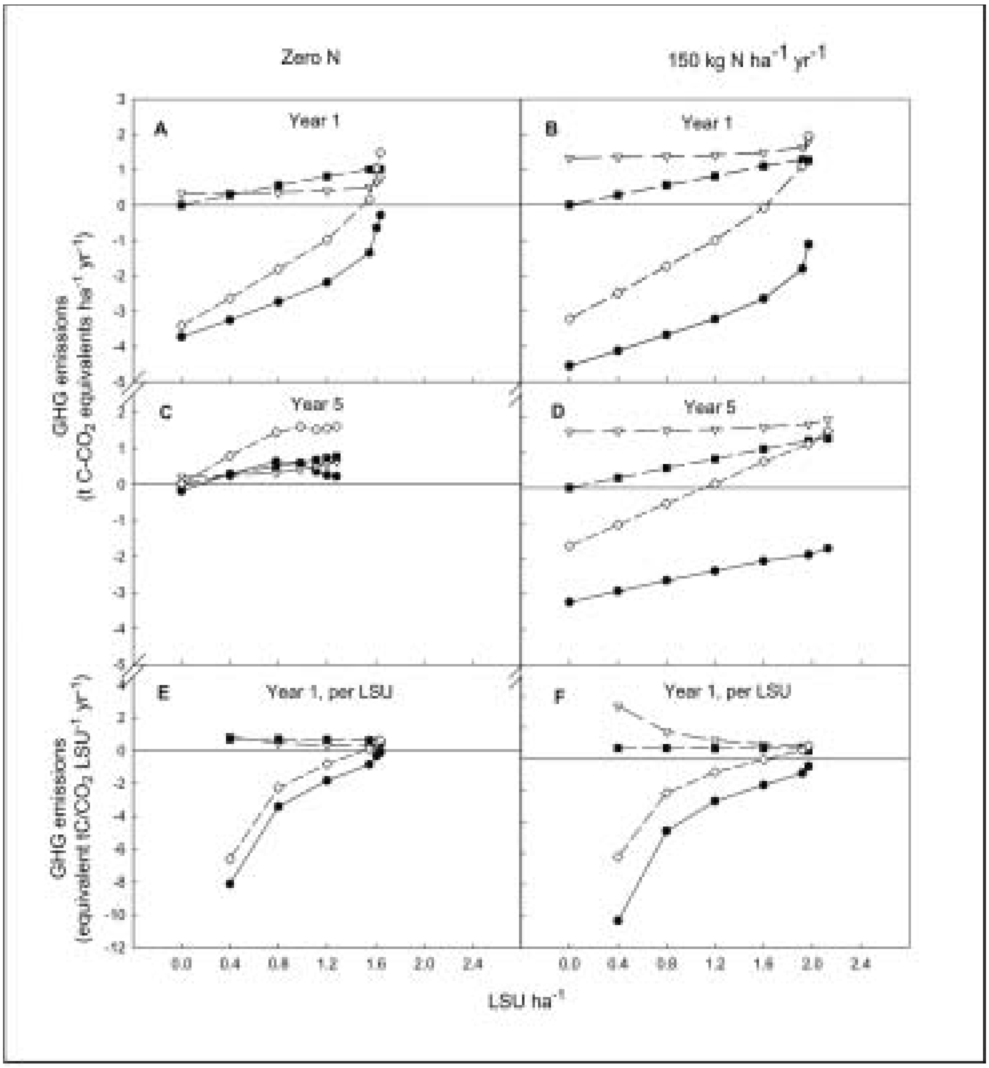


Figure 11 - *Simulated net greenhouse gas emissions and their balance in CO_2-C equivalents of permanent grassland continuously grazed by cattle at a range of mean annual stocking rates (in livestock units, LSU, per hectare) during the first (A, B, E, F) and the fifth (C, D) year after grazing started. Two fertilisation scenarios were considered: no-fertiliser (left hand figures) and 150 kg N ha^{-1} yr^{-1} (right hand figures). Emissions were expressed either on a grassland surface area basis (in t CO_2-C equivalents ha^{-1} yr^{-1}) (A, B, C, D) or on a livestock unit basis (in t CO_2-C equivalents LSU^{-1}) (E, F). A positive value denotes a greenhouse gas source to the atmosphere and a negative value denotes a sink. CO_2 (λ), N_2O (∇), CH_4 (ν), greenhouse gas balance (◊).*

• 4. GHG balance at the farm scale

4.1. Budget at farm-scale

The highest emissions of methane and nitrous oxide are typically seen from intensive ruminant livestock systems. The methane from enteric fermentation in the ruminants contributes greatly to this, even though intensive production with high proportion of concentrates in the feed tends to reduce the emissions per kg milk or meat produced. Indirect emissions of nitrous oxide from leached nitrogen and ammonia volatilisation are often lower for grazing systems compared with housed systems, which also often involve other types of forage crops. Inclusion of grasslands, in particular permanent grasslands, will increase the soil carbon storage. Intensive livestock systems with pigs and poultry may also have high emissions of nitrous oxide, in particular from the fertiliser and manures being applied in the production of the feed for the animals. In addition the manure management systems in intensive pig production is often based on slurry, which can give high methane emissions during storage.

4.1.1. Methodologies for calculating GHG budget at farm scale

Effective mitigation option strategies can only be developed within a whole farm approach It ensures that interactions between the carbon and nitrogen cycles are taken into account. Moreover, farmers will more readily adopt mitigation strategies if these are tailored to their specific farming system. Calculating the GHG budget at farm scale therefore requires a full accounting approach in which the direct and indirect emissions of all GHG as well as the emissions of ammonium and nitrate are accounted for (Figures 12, 13 and 14, Table 10).

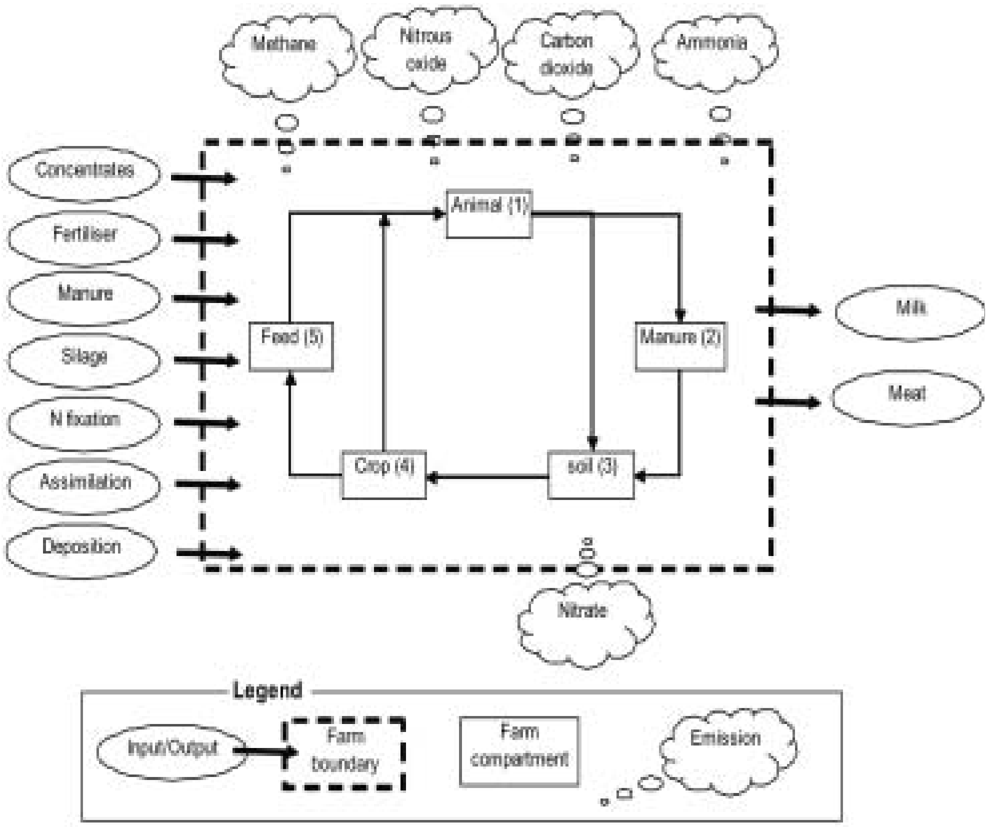


Figure 12 - *Conceptual model of carbon and nitrogen flows of a dairy farm (Schils et al., 2004)*

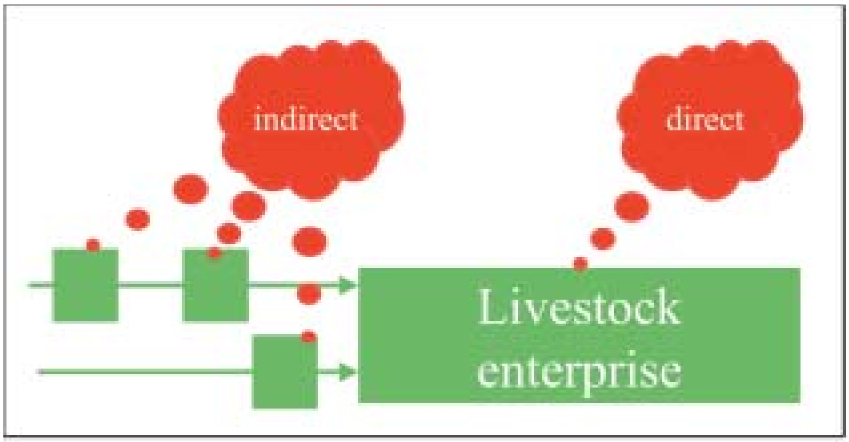


Figure 13 - *Full accounting approach of GHG emission at the farm level*

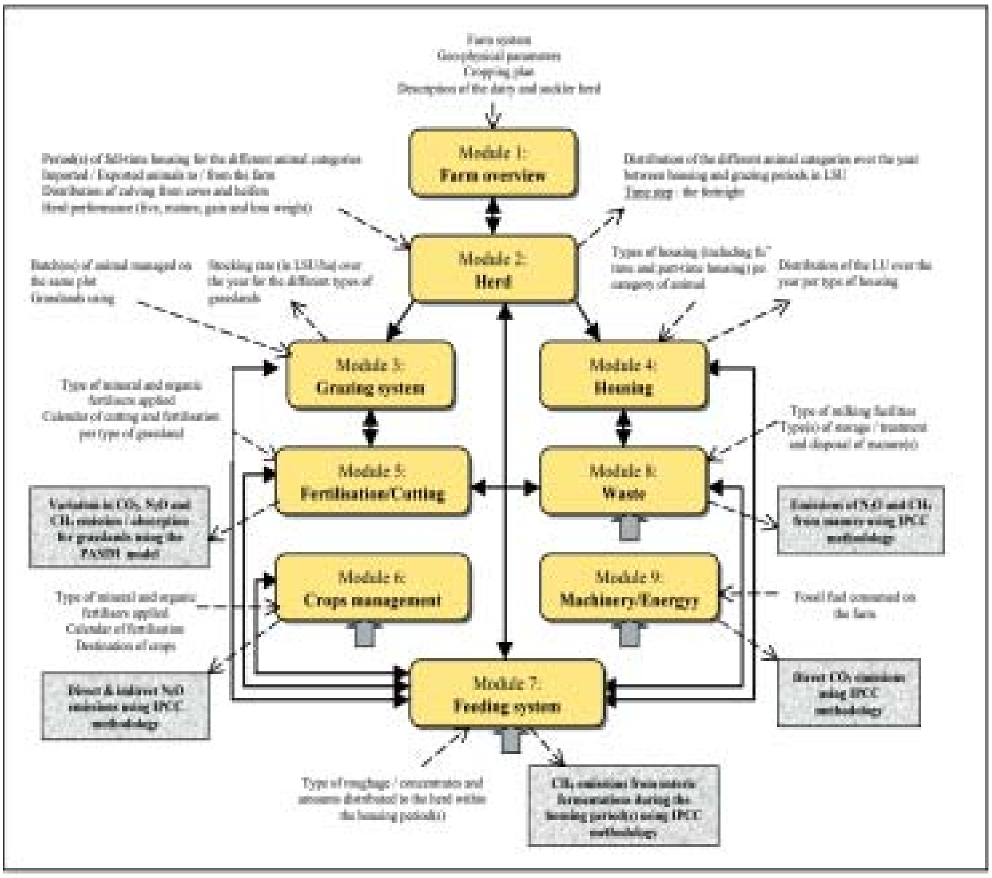


Figure 14 - *Required inputs to assess the GHG budget at the farm level*
Based on the diagram of the FARMSIM model (Salètes et al., 2004): carbon and nitrogen fluxes on a dairy farm. Emissions of GHG at the farm level are calculated using two methods: the standard IPCC methodology (IPCC, 1997) which can be improved with additional sources and emissions factors based on country specific values, and the pasture ecosystem model PASIM for net emissions of grasslands.

Table 10 - *Overview of direct and indirect sources of GHG emissions at the farm level*

Gas	Source	Direct	Indirect
Methane	Enteric fermentation	X	
	Manure management	X	
	Soil	X	
Nitrous oxide	Manure management	X	
	Soil	X	
	Fertiliser	X	X
	Manure application	X	
	Grazing	X	
	Biological fixation	X	
	Silage	X	
	Rumen	X	
	Fuel	X	
	Nitrate leaching		X
	Ammonia volatilisation		X
	Purchase silage		X
	Purchase concentrate		X
	Electricity		X
	Indirect energy use		X
Carbon dioxide	Fuel	X	
	Sequestration	X	
	Fertiliser		X
	Purchase silage		X
	Purchase concentrate		X
	Electricity		X
	Indirect energy use		X

4.1.2. Simulation results at the farm scale

Some recent studies evaluated the GHG emissions from dairy farms using a whole-farm model. (Olesen *et al*., 2004; Schils *et al*., 2004; Salètes *et al*. 2004).

4.1.2.1. A case study: a mixed farming system in Eastern France

Salètes *et al.* (2004) calculated the GHG emission for dairy farm in Lorraine, north-east of France (100 ha with a large milk quota of 300 000 litres) using the FARMSIM model. FARMSIM calculates inputs needed to run the pasture simulation model PASIM (Riedo *et al*., 1998; Riedo *et al*., 2000; Schmid *et al*., 2001) for each of the grassland plot in the farm. The PASIM model allows to simulate the average net annual balance of greenhouse gases (CO_2, N_2O, CH_4) exchanged by the managed grassland plots. Other outputs from FARMSIM are used to estimate CH_4 & CO_2 emissions in the cowshed, according to liveweight, feed composition and level of feed intake, and CH_4 & N_2O emissions from the manure management using the IPCC methodology – Tier 2 (IPCC, 1997).

The studied farm produces milk and steers. Permanent grasslands are the major land use with 76 ha (Figure 15a). In addition, 24 ha are cultivated with a maize-wheat rotation (fallow of 3 ha). The grasslands are grazed from mid April to late October or during part of this period. 32 ha are harvested as spring grass silage in mid May and 9 ha as hay in June. Some pastures are cut as aftermath during summer or as grass silage in autumn. Mineral N is supplied from early March to mid summer to most of grazed and cut grasslands. Intensive (40 ha) and semi-improved (36 ha) grasslands are spread with

110 and 45 kg N ha^{-1}, respectively. In addition, the intensive grasslands receive in autumn 12 tons long term stored manure per ha (i.e. 70 kg N ha^{-1}). Other grassland types get by mean 60 kg kg N ha-1 from fresh manure in winter.

The dairy herd consists of 48 Prim'Holstein cows producing about 6700 kg milk per year. The roughage balance of this grazing livestock farm is rather tense with a global stocking rate of 1.4 LSU ha^{-1} relying on 13% maize crop, 46% intensive grasslands and 41% semi-intensive ones. Figure 15bcd shows some output charts about the variation of the herd demography (indoor vs. indoor), stocking rate & feed supply over the year after compilation of all these farm data using FARMSIM.

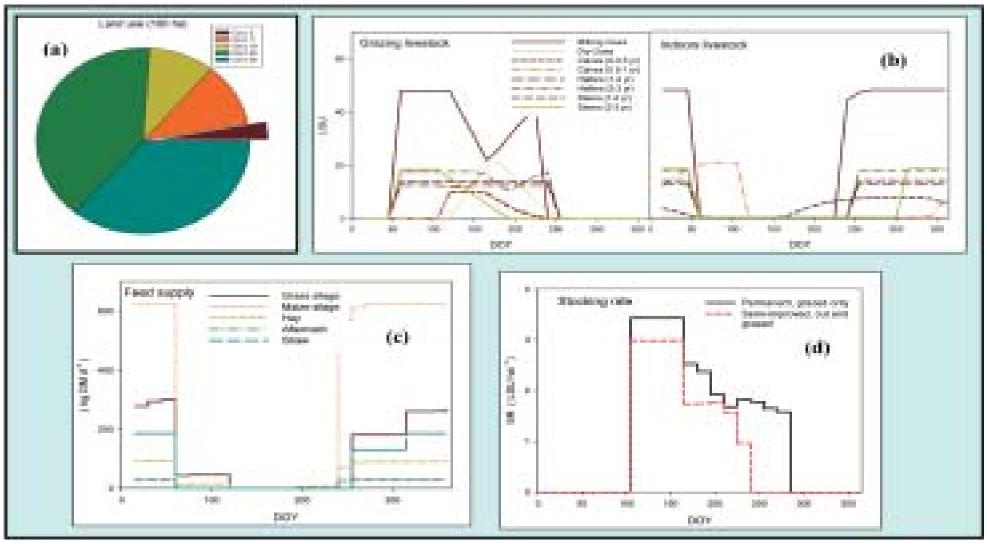
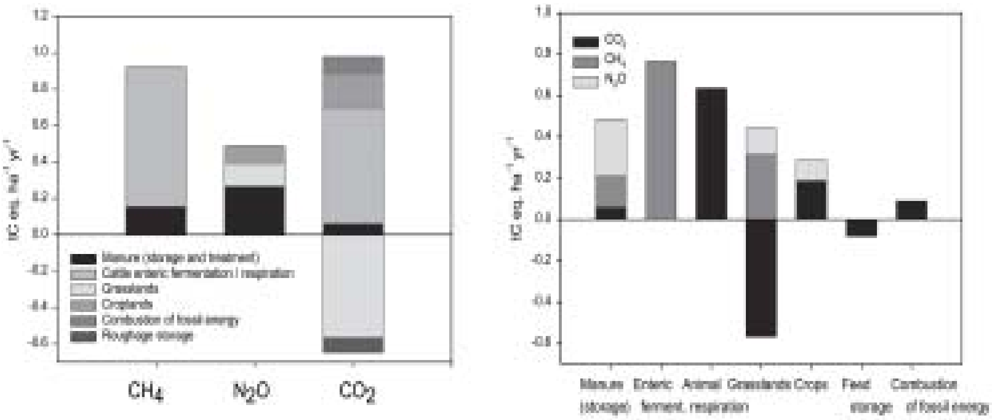


Figure 15 - *Land use (a) and variation of the herd demography (indoor vs. indoor) (b), stocking rate (c) & feed supply (d) over the year for the studied mixed farming system in Eastern France*

Results of the first simulations are presented in Figure 16. The farm emits as a whole about 1.8 t C eq ha^{-1} yr^{-1}. Emissions of CH_4 from enteric fermentation (either indoor or outdoor) and CO_2 from cattle respiration are a big source in the total farm GHG budget. CO_2 emissions from combustion of fossil energy is relatively small compared to the other sources of CO_2 (fluxes from croplands and composting process & respiration from animals). N_2O emissions from manure management is possibly overestimated due to lack of precise reference in France on emission factors for solid wastes.

Figure 16 - *GHG balance simulated with the FARMSIM model for a mixed dairy and beef farm of 100 ha in Eastern France. After Salètes et al. (2004)*



4.1.2.2. Comparison of GHG emissions from a white clover-based dairy system with a grass/fertiliser-N system

Schils *et al.* (2004) calculated the GHG emissions for a well described dairy farm on clay soil. The farm consisted of 41 and 34 ha of grasslands of which 55% had been renewed in the last three years. The N application (mineral fertiliser or manure) amounted respectively to 69 and 275 kg N ha^{-1} yr^{-1}. The herd consisted in 59 cows on both farms, with a stoking rates respectively of 2.2 and 1.9 LU ha^{-1}. The herd produced 8095 and 8294 kg milk cow^{-1}. The biological fixation was estimated to 176 kg ha^{-1} yr^{-1} for the white clover-based dairy system. The amount of concentrates given to both herd was around 1820 kg cow^{-1} yr^{-1}. Figures 17 & 18 present respectively the estimated nitrogen and carbon on this dairy farm, and the direct and indirect GHG emissions at the farm scale.

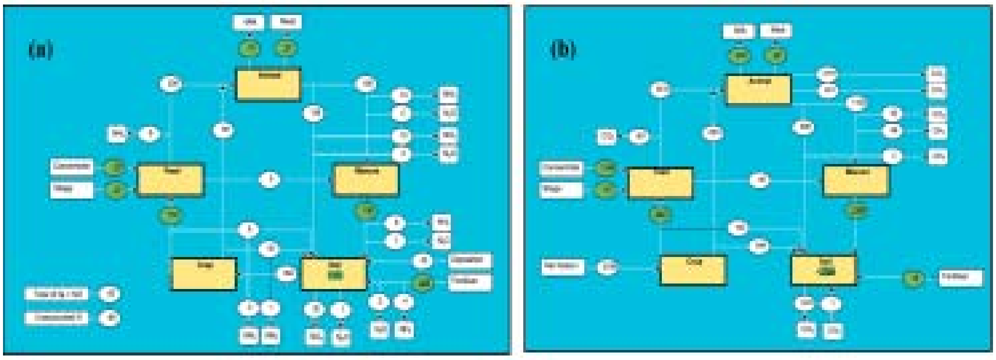


Figure 17 - *Nitrogen (a) and Carbon (b) flows (kg/ha) - After Schils et al. (2004)*

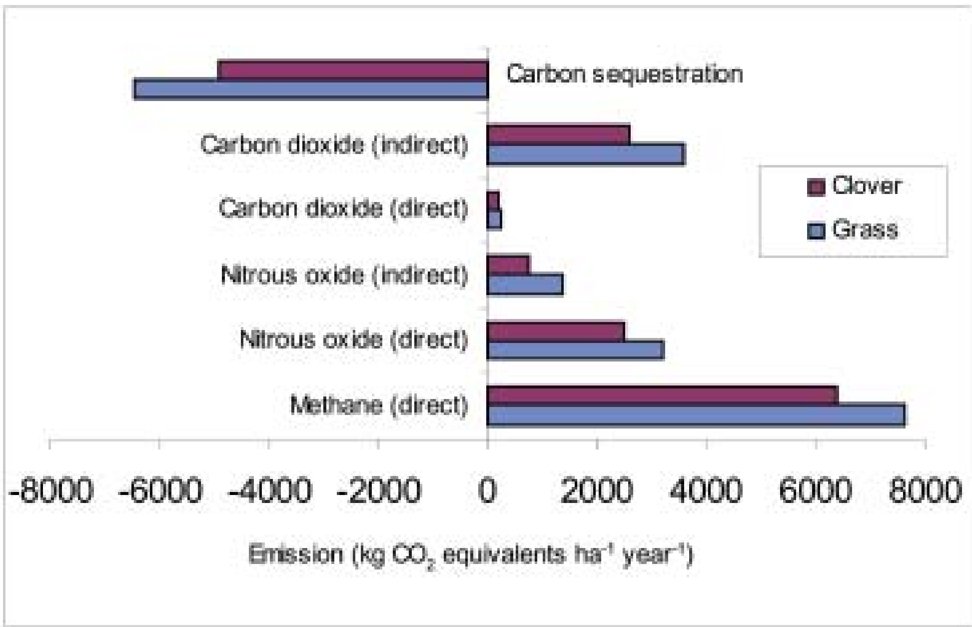


Figure 18 - *Direct and indirect GHG emissions from a white clover-based dairy system and a grass/fertiliser-N system*

Table 11 - *Effect of mitigation options on direct and indirect emissions on grass/fertiliser-N farm. Greenhouse gas emissions of present situation are expressed in CO_2-equivalents per ha. Results of scenarios are relative to"present". After Schils et al. (2004)*

	Present	Less mineral fertiliser	Less grazing	More milk per cow	No grassland renovation
Methane	7623		289	-281	
Nitrous oxide (direct)	3220	-151	-273	-73	-10
Nitrous oxide (indirect)	1381	-34	-29	-88	24
Carbon dioxide (direct)	263		55	10	-10
Carbon dioxide (indirect)	3582	-82	-5	-11	-4
Carbon sequestration	-6468				-843
Total (kg CO_2 ha^{-1})	9597	-267	37	-443	-843
Total (kg CO_2 kg $milk^{-1}$)	0.70	-0.02	0.00	-0.03	-0.06
Ammonia volatilisation (kg N ha^{-1})	57	-0.6	3.4	-1.1	0
Nitrate leaching (kg ha^{-1})	20	-1.5	-1.8	-2.1	0

The direct and indirect emissions presented in Figure 18 and Table 11 are according to the overview in the effects of the mitigation options are specific for these farming systems under the local conditions. However, the results demonstrate that a single measure can affect the emission of other GHG and also other environmentally relevant emissions like nitrate and ammonia.

Comparison of GHG emissions from European conventional and organic dairy farms

Olesen *et al.* (2004) developed the FarmGHG model that predicts carbon and dairy flows on dairy farms and design with the aim of allowing a quantification of all direct and indirect gaseous emissions from dairy farms (Olesen *et al*. 2004). The imports, exports and flow of all products through the internal chains on the farm is modelled. FarmGHG thus allows assessments of emission from both the production unit and all prechains. The model includes N balance and allows calculation of environmental effect balances for GHG emissions (CO_2, CH_4 and N_2O) and eutrophisation (nitrate and NH_3).

Model dairy farms were defined for five European agro-ecological zones for both organic and conventional systems. The model farms were all defined to have the same utilised agricultural area (50 ha). Cows on conventional and organic model farms achieve the same milk yield, so the basic difference between conventional and organic farms was expressed in the livestock density. The organic farms were defined to be 100% self-sufficient with respect to feed. The conventional farms, on the other hand, import concentrates as supplementary feed and their livestock density was defined to be 75% higher than the organic farm density. Regional differences between farms were expressed in the milk yield, the crop rotation systems, the cow housing system and manure application method common to the region.

The model results show that the emissions at farm level may be related to either the farm N surplus or the farm N efficiency. The farm N surplus appears to be a good proxy for GHG emissions per unit of land area. The farm N surplus can relatively easily be determined on practical farms from the farm records of imports and exports and from the composition of the crop rotation. The GHG emissions per product unit (milk or metabolic energy) were quite closely related to the farm N efficiency. This farm N efficiency may therefore be used as a proxy for comparing the efficiencies of farms with respect to producing products with a low GHG emission.

4.1.3. Knowledge gaps

These farm approaches have revealed some knowledge gaps, which have been discussed at the workshop:

- How much of imported feed (or C) ends up as SOM? Does it depend on soil conditions, soil type or management?
- Which practice is more cost effective: sequester additional organic carbon in C depleted soils or conserve existing C stocks in soils which are already rich in organic C?
- Manure to arable and fertiliser to grass? Is that really the best option always?
- Reduced methane from increasing concentrates in diets to cattle may increase nitrous oxides emissions elsewhere in the system. What is the balance?
- Effectiveness and feasibility of measures to reduce enteric fermentation in animals (e.g. breeding, immunisation)?
- Is timing of mineral N application really more critical than for manure application?
- Sustainable and reliable technologies for scrubbing ammonia, methane and nitrous oxide from air from point sources are needed (houses, manure storages), but these should not increase nitrous oxide emissions.

4.2. Mitigation options in agriculture

4.2.1. Technical issues and decision making at farm level

4.2.1.1. Animal husbandry

There are numerous techniques which are currently being evaluated to reduce the methane emissions from enteric fermentation (Table 12). These techniques may have a large potential, but much more research and development is still needed. Some of these techniques may also not be acceptable to consumers, especially after the Bovine Spongiform Encephalopathy (BSE) crisis in Europe.

Table 12 -*Mitigation options and animal husbandry*

Option	Effect	Side effects	Costs	Overall evaluation
Increase concentrate proportion in feed ration	Reduced for CH_4, but small when other gases are considered	Reduced grassland proportion, limits for intensive systems	Realistic	(+)
Rumen by-pass ingredients	?	Animal changes	High	(+)
Feed additives	?	Many of them are not acceptable	Realistic	(+) R&D
Genetic selection	?	Feasibility?	?	R&D
Increasing number of lactations	<5-10%	Requires feeding with forage crops	Realistic	(+)
Slaughter beef cattle at younger age	Small	Consumer demand?	?	-
Spring calving	?	Conflicts with constant milk production, linked with grazing and feeding practice	?	-
Reduce N surplus in diet (e.g. synthetic amino acids, balanced protein diet)	Reduced N_2O emission	Reduce NH_3 emissions	Realistic	++

4.2.1.2. Housing

The frequent removal of manure from house and a reduced exposed area in new farm buildings may offer some mitigation options to be considered (Table 13).

Table 13 - *Mitigation options at the housing scale*

Option	Effect	Side effects	Costs	Overall evaluation
Slurry based system				
Reduce exposed area (change in floor structure, cleaning floor, oil cover)	reduced NH_3 and N_2O	Better climate in house	Realistic for new houses	+
Frequent removal of manure from house	reduced CH_4 and NH_3	Better climate in house	Realistic	+
Cooling of manure	reduced CH_4 and NH_3	Energy consumption, better climate in house	High (cattle) Realistic (pigs)	(+)
Treatment of air (mechanical ventilation)	reduce NH_3, CH_4 (?) N_2O (?)	Not possible in housing systems with natural ventilation for higher animal welfare	High for CH_4 and N_2O	(+) R&D
Acidification	reduce NH_3, CH_4 (?)	Odour problems, safety problems	Realistic for H_2SO_4	(-)
Separate solid/liquid based system				
Frequent removal of solid manure from house	reduced NH_3 and N_2O	Better climate in house	Realistic	0
Deep litter based system for cattle				
Reduce retention time	Reduced $_{CH4}$ (50% reduction)	Labour costs	High	(+)
Deep litter based system for pigs				
Nitrification inhibitor	Reduced N_2O	Increased leaching	High	-

4.2.1.3. Manure stores

The development of anaerobic digestion facilities is of interest and appears to be realistic, especially at the community scale (Table 14).

Table 14 - *Mitigation options and manure stores*

Option	Effect	Side effects	Costs	Overall evaluation
Slurry				
Surface crust (straw layer on slurries that do not naturally form crusts)	Reduce CH_4 (5-15%) and NH_3 (80-90%), but higher N_2O (0.1-1% of incoming N)	Reduced odour	Low	+ R&D
Solid cover	Reduce NH_3 (90%) and CH_4 (0-50%)	Reduced odour	Realistic-high	+
Anaerobic digestion (farm or community scale)	Reduce CH_4 (60-70%), NH_3 (depends on storage) and N_2O. Fossil fuel substitution.	Risk of NH_3 from field application. Reduced odour. Co-digestion solves organic waste problem from other sectors	Realistic	++
Aeration	Reduce CH_4 (90%), increase N_2O (10% of incoming N)	High energy consumption. Reduced odour	High	--
Dilution (double volume)	Reduce NH_3 (20-50%), reduce N_2O (10-20%)	Increase transport costs and storage capacity	Realistic-high	(+)
Solid manure, low dry matter content				
Add straw, woods or other structural material, giving composting	Reduced CH_4 (30-70%). Increase in N_2O (from 1% to 2% of N input, doubling). Increase in NH_3	Reduced odour, solve organic waste problems from other sectors, good product	Realistic to profitable	+
Solid manure, high dry matter content (e.g. deep litter)				
Compaction	CH_4 (??), Reduced NH_3, Reduced N_2O	Increased NH_3 emission during field application	Low	-
Treatment of air from Composting	NH_3 (90%), CH4 (?), N_2O (?)	Energy consumption, reduced odour	High	(+)

4.2.1.4. Other sources and sinks

Table 15 - *Mitigation options for other sources and sinks*

Option	Effect	Side effects	Costs	Overall evaluation
Clean walkways and collection yards	Reduce NH_3, CH_4 and N_2O	Labour intensive and partly infeasible	High	-

4.2.2. Elements, issues and decisions from the farmer's perspective

Estimates for the net exchange of these greenhouse gases between agriculture and atmosphere are uncertain, because sources and sinks are diffusively spread over the countryside and influenced by environmental factors which greatly vary in space and time (Oenema *et al*., 2001, Kuikman *et al*., 2003, Oenema *et al*., 2004). The uncertainty in the estimates and the variability in controlling factors not only frustrate the accurate accounting of greenhouse gas emissions, they also obstruct the successful implementation of policies and measures that aim at mitigating emissions of CH_4 and N_2O from agriculture.

The success of such policies largely depends on the response of farmers to these policies (and to other policies as well as to developments in markets and technology). So far, the role of the farmer is often overlooked in the estimation and mitigation of greenhouse gas emissions from agriculture. Policies need to consider not only the best-available scientific information but also the role of farmer as well.

The farmer combines the functions of entrepreneur, manager and craftsman. The skills for these functions and personal characteristics (preference, ambition, vision, responsibility) greatly differ among farmers. Farm characteristics (type, size, intensity) and environmental conditions (climate, morphology, soil, and hydrology) differ also widely in for example the EU-15. The combination of personal and farm characteristics and external conditions (market developments, policies, contractors, suppliers, processing industry, extension services, farmer unions, neighbours and family) determine the decision environment of the farmer. The decision environment in turn determines farm performance, and whether or not mitigation measures are implemented, and how and to what extent (Figure 19).

Options for mitigation at farm level can be categorised into (i) management measures, (ii) technological measures, and (iii) structural measures, as done by Velthof *et al.* (2004) in the decision support system MITERRA-DSS. Management measures focused on improving resource use efficiency (e.g. energy, fertilisers, water, animal feed) are the most practical in the short term (Table 16). There is also great scope in improving resource use efficiency in agriculture in a cost-effective way, as follows from for example differences in performance between farms (Burczyk et *al*., 2001; Vellinga *et al*., 2004; van Groenigen *et al*., 2004). For example, due to the implementation of the so-called manure policy, N use efficiency at farm level in The Netherlands increased on average by a factor of 2 between 1998 and 2003, and as a result N_2O emissions decreased by about 30%. Farmers require proper (improved) management skills to render such improvements cost-effective. Personal guidance of farmers, extension services and demonstration farms appear all very instrumental in improving management skills. Further improvements in N use efficiency are anticipated, yet may result in trade-offs and not always lead to positive results for all environmental policies. Implementation of technological measures often requires investments (e.g. manure digesters) but these may be cost-effective too in the medium term. Structural adjustments, e.g. decreasing the volume of production and number of animals, changing crop type or transferring production from one country (area) to another) are expensive, and only cost-effective in terms of mitigation of greenhouse gas emissions in the longer term. Recent structural adjustments in The Netherlands by the government (buy-out of animal rights, and lowering the milk quota) have decreased N_2O emissions

about 5% and CH_4 emissions about 1%.
Summarising, farmers are crucial in the mitigation of greenhouse gas emissions from agriculture. Improving resource use efficiency is the most cost-effective measure for mitigating greenhouse gas emissions in the short term. Indeed, there is great scope for improving resource-use efficiency in practice. Success in terms of less greenhouse gases will be determined by the willingness and skilfulness of individual farmers and by elimination of negative trade-offs with other emissions to air and water (Oenema *et al*., 2004)

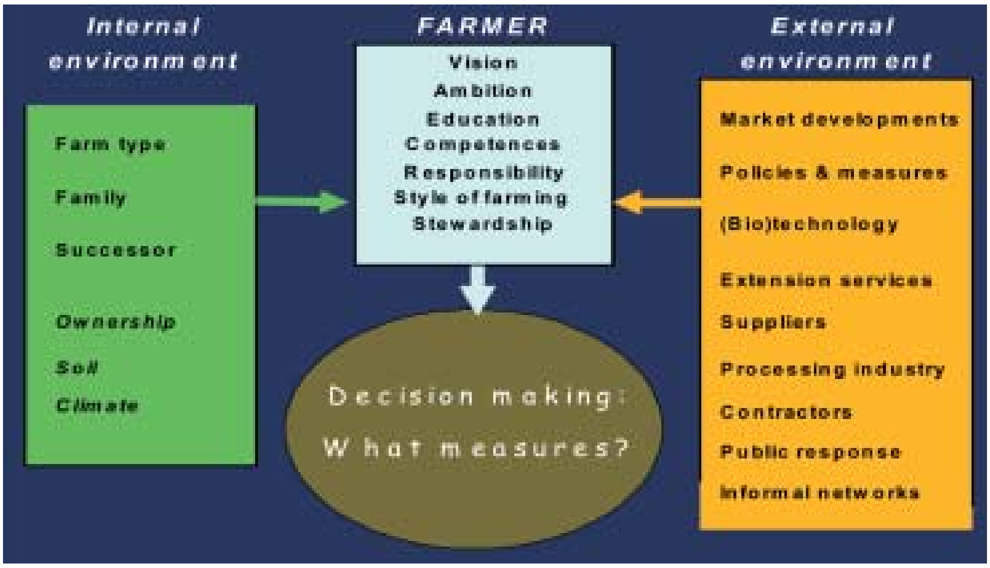


Figure 19 - *Decision environment of farmers (Oenema et al., 2004)*

Table 16 - *Mitigation emissions at farm level (Oenema et al, 2004).*
"Button" refer to factors which can be altered by the actions taken by farmers.

	The N Button	The C button	The Water button
Management	Fertilisation Soil cultivation grazing	Manure storage Soil cultivation grazing	Irrigation Drainage Groundwater level
Technology	Fertiliser type Manure processing Housing system	Manure digestion Soil cultivation	
Structural Change	Farm type Animal number Other crops	Farm type Animal number Other crops	Flooding Water buffers

Currently, national inventories use a top-down approach, aggregated information on activities is multiplied with an emission factor. It is input-based, and therefore only effect of changes in inputs show up. Efficiency improvement do not affect the national inventory, if they are not accounted for by changes in emission factors. A bottom-up approach, i.e. farm approach, could be an incentive for the stakeholder. Indeed, mitigation options have to be taken by a farmer, requiring an insight into the trade-offs with other emissions. Furthermore, a whole farm approach allows for the wide variation that exists in farming practices.

Figure 20 - *The effect of increasing pressure by economic instruments on human behaviour. After Von Weizsacker & Jesinghaus (1992)*

In conclusion, farm decision making can be oriented towards GHG mitigation in several ways :

1. Improve skilfulness of farmers through education and communication;
2. Get Climate Change and mitigation to the farmer both as concept and perception;
3. Active participation of farmers start with recognising emissions and C stock changes and that credits and accounting is possible;
4. Develop packages of best management practices that fit distinguished styles of farms;
5. Optimise horizontal organisation of farms (consider conglomerates);
6. Optimise vertical organisation of farms (from resource supplier to processing industry, retailers and consumers);
7. Optimise dialogue between farmers, policy makers and scientists and the public (consider the image of farming as a sound basis for good farmers).

4.3. Policy level

There are several drivers which influence changes in agricultural production and which may, indirectly, influence GHG emissions by livestock farms (Table 17). Changes in livestock farming systems may offer some possibilities to mitigate GHG emissions, as indicated from a case study in France (INRA, 2002; Soussana *et al.,* 2004b).

In France, farms raising domestic herbivores occupy two-thirds of farmland, and 60% of all professional farms raise some herbivores (Bontron *et al*., 2001). However, grasslands, which occupied about 25% of the total land area in 1970, have declined markedly in favour of arable, including fodder crops such as maize silage and fallow. Restoration over 20 years of half the amount of land under permanent grass lost since the 1970s would give a mean annual increase of 90,000 hectares, estimated to lead to an increase of 16 Mt of soil organic carbon during this period (INRA, 2002). This is, however, equivalent to only 10% of the annual CO_2 emissions from fossil fuels in France. Moreover, such an increase in the grassland area would imply major changes to livestock breeding systems and grassland management. In addition, the consequences on the emissions of other greenhouse gases (CH_4 and N_2O) are still unknown (Soussana *et al*., 2004b).

Another possibility concerns the extensification of livestock farms. Herds of domestic herbivores are still generally feeding on grasslands, which occupy more than 80% of all land put in fodder crops (15% for maize silage). However, the proportion of grass differs considerably as a function of animal production system: it reaches nearly 95% in cattle farms which breed for meat production, but is much smaller in, for example, the 49,000 dairy farms which produce half of all French milk and 20% of French meat using much more intensive production systems (with 41% of maize silage and a stocking rate of 1.7 LSU ha^{-1}) (Bontron *et al*., 2001). The extensification of intensive livestock production provides an interesting option to enhance soil carbon accumulation by increasing the proportion of grass in the diet. This would involve conversion of annual fodder or cereal crops into temporary grassland and conversion of temporary grassland into permanent grassland (Soussana *et al*., 2004b).

Such extensification could also reduce the CH_4 emissions per unit land area, because of a lower stocking rate, and the N_2O emissions through the limitation of nitrogen inputs. However, it is necessary to take account of the emission coefficient associated with symbiotic nitrogen fixation by leguminous crops, which current estimates (IPCC, 1996a) equate with the coefficient for nitrogen fertiliser. From an agronomic point of view, several options can be proposed to increase and optimise the use of grass in livestock breeding systems: an increase in the length of the grazing season, the use of deferred grazing, lowering the costs of production by using grass-legume mixtures or permanent grassland and more efficient use of livestock manures.

At the policy level, mitigation measures are perhaps best encouraged as part of a broader environmental agenda. Smith & Powlson (2003) and Smith (2003a, 2004a) argue that GHG mitigation needs to be tackled hand in hand with other related problems. For example, the IPCC (2001a) have noted that global, regional and local environmental issues such as climate change, loss of biodiversity, desertification, stratospheric ozone depletion, regional acid deposition and local air quality are inextricably linked. GHG mitigation clearly belongs on this list. The IPCC (2001a) further noted that recognising the linkages among environmental issues, and their relationship to meeting human needs, provides an opportunity to address global environmental issues at the local, national and regional level in an integrated manner that is cost-effective and meets sustainable development objectives. The importance of integrated approaches to sustainable environmental management is becoming ever clearer (Smith, 2003b).

Though there are often co-benefits of GHG mitigation measures (e.g. positive effects on biodiversity, erosion control, fertility, soil moisture), there may also be conflicts. For example, extensive areas that are managed for biodiversity may have low soil carbon values (but see Falloon *et al.*, 2004 where the opposite is the case), and agricultural areas placed under long-term GHG mitigation management may reduce the adaptive capacity of the agricultural sector. As discussed earlier, one must also consider the trade off between GHGs and other implications for fossil fuel use (e.g. pre-chain fossil fuel use in fertiliser and herbicide production (see Frye 1984; Lal, 2004), and fuel carbon costs for transport, crop drying and processing and field management practices (e.g. Smith & Smith; 2000; Lal, 2004).

Wherever possible «win-win» options (whereby benefits accrue through other means, e.g. increased fertility or production) should be targeted (Lal *et al*., 1998; Smith, 2003a) as should «no regrets» options (whereby the management practice yields immediate benefits as well as potentially in the future; Smith & Powlson, 2003; Smith 2004a). In addition to attempting to solve several environmental problems together, social and economic problems also need to be addressed in the same package. All of the scientific and technical measures outlined in this paper have the potential to enhance C sinks, but the extent to which these are sustainable also needs to be considered (Smith 2003b). In the context of the current changes in the Common Agricultural Policy, the integration of GHG mitigation in the framework of agri-environmental policies is likely to be more attractive

The renewed and actualised CAP of the EU (Council regulation EC n° 1782/2003) makes grasslands nowadays more attractive for farmers than before. Article 5.2. of this Regulation obligates Members States to ensure all that land which was under permanent pasture at the date provided for the area aid application for 2003 to be mentioned under permanent pasture (cited by Carlier *et al*., 2004). In annex IV the standard for “Good agricultural and environmental conditions” (GAEC) are mentioned with special attention for permanent pasture. The Member States themselves must introduce their own policy in the development of these GAEC, so that the farmers will receive direct payments in return for their responsibilities towards protection of the environment, animal health and welfare and public health (so called “Cross Compliance”). The development of such measures in combination with the technical and farm management issues discussed above may offer a realistic way to mitigate the GHG emissions from livestock farms.

Table 17 - *Drivers of change affecting agricultural production and greenhouse gas emissions*

Driving force	Pressure	Result	Effect on GHG emissions
Increasing population	Intensification		Increase
EU CAP, agricultural policy, WTO	Maintain income	Migration, increase farm size, reduce cost	Decrease for CH4 and N2O. C-storage??
Prosperity, consumer choice of food, food safety	Quality of produce, more diverse range of products, flexibility, higher meat consumption	Increasing local production, regionalised food production, need for and risk in specialisation	Depends
Other EU regulations, e.g., NEC directive (emission ceilings for ammonia and other pollutants), nitrate directive	decrease air and groundwater pollutants		Decrease N_2O as co-benefit (nitrate dir.); NEC: possibly increase direct N_2O from soils, decrease indirect N_2O emissions from agriculture
Other values (animal welfare, environment, landscape)	Negative image of agriculture, Diversification, Regulations	Fewer skilled farmers, changes in management systems	Depends
Other claims for land (urban, infrastructure, afforestation)	Intensification		Decrease
Climate change	Adaptation, extreme events, water resources	Land use change (northward shift), modification of environmental impacts of agriculture (SOM mineralisation), expand production areas for sensitive crops, regionally specific effects	Increase for CH4 and N2O (CO2-effect uncertain)
Need for clean water and environment	Restrictions on land use	Extensification regionally, higher resource use efficiencies	Decrease (some technologies may increase emissions)
Energy price (increase)	Produce energy, reduce energy cost (direct and indirect, non-factor input)	Reallocation of production, development of energy efficient technologies, bioenergy crops	Decrease
Labour and skill availability (lack of)	Negative image of agriculture	Fewer skilled farmers	Increase
Research and development	Increasing productivity, capital intensive agriculture		Depends on whether GHG is a driving force in R&D
Economic power of farms	Reduced capital	Less ability for actions, lower recourse use efficiency	Increase
Labour age structure	Aging of land labour	Less flexibility and adaptability	Increase

• 5. Development of spatially explicit baseline and alternative scenarios of GHG emissions in Europe

5.1. Estimates of current emissions

5.1.1. Calculation of management input parameters

Developing runs of PASIM at the European level requires an upscaling procedure. The model, by itself, does not require upscaling so far because it is mechanistic and so highly generic. The regional dependencies are mainly due to the input parameters and variables. Therefore, the main work which has been done for this upscalling within the framework of the GREENGRASS project consisted to obtain the set of information required by PASIM in input at the level of the Europe. With this set of input data, PASIM has ran at a spatial resolution of one degree until the equilibrium state, which means that PASIM has ran as long as the different carbon and nitrogen pools change over a year-period. Tables 18 and 19 present the climate and soil data that have been used as inputs and the way management has been implemented at each grid point according to different scenarios (Figure 21).

5.1.2. Results

CO_2

Simulated NEE over Europe is presented on Figure 22 (Vuichard *et al*., 2004). Values are always negative as expected for an equilibrium run of a scenario with management. These values are equal to the opposite of all the carbon that goes out of the system (i.e. yield and animal C assimilation (milk, liveweight increase). Values are ranged from 0 to 5.3 t C ha^{-1} yr^{-1} with maximum values on Atlantic coast (West of France, North of Spain) and Italy. Lowest values are obtained for high latitude regions as Scandinavia or Iceland.

N_2O

Simulated N_2O emissions by PASIM are presented on Figure 23. Values are ranged from 0 to 10 kg N ha^{-1} yr^{-1}. Emissions on grazed grasslands are higher than those on cut grasslands (Figure 24) because of the animal's dung and urine that go back to the soil. The main driver of N_2O emissions is the annual amount of N-fertiliser.

CH_4

Figure 25 presents the methane emissions defined by PASIM with values ranging from 0 to 160 kg CH_4-C ha^{-1} yr^{-1}. Values are only representative of emissions on grasslands (e.g. when animals graze) and are mainly driven by the animal density defined in each region.

GHG total flux

Expressing these results in term of GHG total flux requires to convert emissions per species in CO_2-equivalents by multiplying them with their global warming potential (IPCC, 1996). CO_2 contribution to the total GHG flux is the Net Biome Productivity (NBP). However, since PASIM was run to the equilibrium state, the NBP is by definition nil. If the estimate of a NBP of 101 TgC per year (Janssens *et al*., 2003) over European grasslands is made, then this NBP can be distributed among grid points in proportion to the simulated NEE. Distribution of this NBP over Europe is shown in Figure 26. By using this estimate for NBP, total GHG fluxes can be recalculated (Figure 27). Figure 28 shows relative contributions of methane and nitrous oxide compared to CO_2. and shows that contributions of N_2O and CH_4 increases in parallel to the carbon sink. There is a cross-point which separates the graph: above this cross-point, the grasslands are sources of GHG, below the grasslands are sinks of GHG. This first European scale simulation of the GHG balance of grasslands will be further improved: first by developing alternative management scenarios and second by developing algorithms to estimate the management of grasslands from the data available in the European statistics for agriculture.

Table 18 - *Origin of data used for running the PASIM model*

Meteorological drivers	For climate data, high-resolution climate from the ATEAM European project (combines a 10 minutes resolution monthly climatology with the 1900-2000 monthly anomalies from the CRU) and ERA 15 ECMWF reanalysis (for missing parameters). To simulate the daily and hourly variability we used the variability observed in the ECMWF reanalysis
Soil data	Texture data come from a global dataset of soil types available at 1-degree latitude by 1-degree longitude resolution. Soil types are described at two levels of detail, including 106 types based on Zobler's assessment of FAO Soil Units and an aggregated list of 27 types at the Great Group level. Field capacity, wilting point and bulk density come from the "Global Gridded Surfaces of Selected Soil Characteristics (IGBP-DIS)" data set. All the surfaces are global, at a resolution of 5x5 arc-minutes
Land cover information	CORINE land cover map which provides 44 types of vegetation including grassland with high resolution (250m) and it based on analysis of LANDSAT and SPOT images over Europe. For regions on which CORINE was not available, the PELCOM land cover map was used
Management information	PASIM requires management parameters as input data. They consist in: ➢ dates of harvest ➢ animal stocking rate and grazing periods ➢ dates of application and amount of N-fertilisers However, this set of information is not available at the European level and there is a wide range of different practices at the level of a grid point (it may be not suitable to run PASIM with spatial mean values for these parameters). A set of rules that aims to specify automatically values for the management parameters has been developed. This set of rules tends to optimise the grassland use.

Table 19 - *Processing implemented to define the management drivers in each grid point*

Fraction of cutting vs. grazing grasslands and animal stocking rate	We consider that grasslands may be either cut or grazed by animals. Thus, two set of runs have been performed: One for which grasslands were grazed, another one for which they were cut. This requires defining the proportion of cut grasslands and grazed grasslands on each grid point. This management parameter has been defined with an optimal algorithm. This algorithm (Figure 21) tends to maximise the animal density in each grid point under the constraint to keep each grid point self-sufficient in term of animals feed production. The run of the cutting scenario enables to know the yield production (Y kg per ha of cut grasslands) on offer in each grid point. The run of the grazing scenario with a certain animal stocking-rate (number of animals per ha of grazed grasslands) in input provides the number of days during which animals have to be out of the grassland and thus, the quantity of fodder required per ha of grazed grasslands (X). With these two variables, we can define the fraction of grazed (F) and cut grasslands link to the stocking rate applied. Multiplying the stocking rate applied with the fraction of grazed grasslands gives the animals density per ha of grassland (D) on each grid point. The maximal animals density is reached by increasing iteratively the animal stocking-rate. Days of harvest are function of plant growth. They occur every 30 days at the most. They are conditional to a positive harvest at that day and to a slight decline of plant growth rate over a 10 day-period.
Grazing	For the grazed scenario, animals may graze if the shoot biomass is higher than a threshold equals to 0.2 DM kg m^{-2}. When shoot biomass is lower than this threshold, animals are out of grassland and fed by yield production from cut grasslands. During these indoor periods, fluxes (respiration, dejection, methane) from animal production are not taken into account.
Timing of N fertilisers application	After each cut (cutting scenario) or every two months (grazing scenario), grasslands may be fertilised. These predefined dates of fertilisation are conditional to an N assimilation by plants at that time and to a soil water content lower than the soil water content at field capacity.
Amount of fertiliser applied	An optimal application procedure has been developed which maximises the production of biomass with the minimal amount of fertilisers. This procedure requires two runs of PASIM per grid point. The first run consists to apply 500 kg N/ha at each application (which tends to develop a N-saturated run) and to store values of carbon allocated into shoot biomass during two applications. The second run allows to define the minimal amount of nitrogen for each application: this enables to allocate the same quantity of carbon into shoot biomass than in the first run for the period that precedes the application. The amount nitrogen defined at the end of this second run is the minimal quantity required to reach the maximal production. Because farmers never try to reach this optimum, we decided to apply 30% of this annual N-amount.

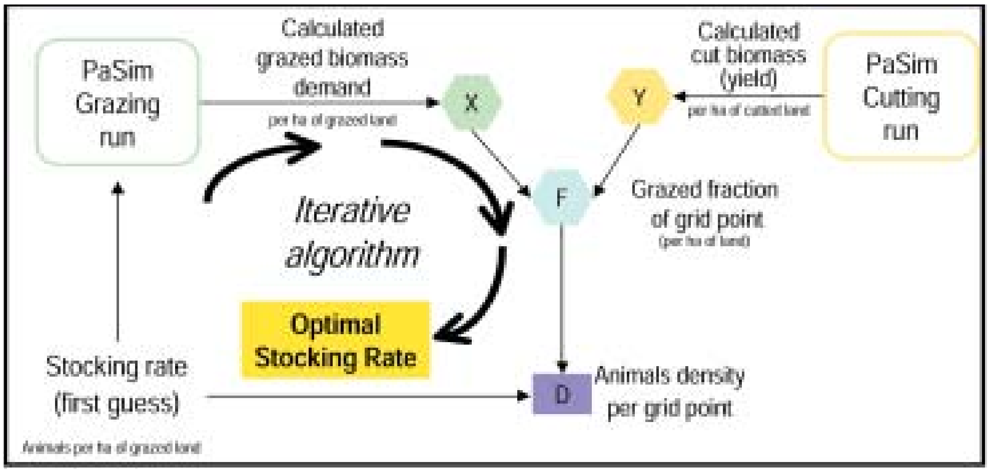


Figure 21 - *Optimal algorithm to define management using the PASIM model (Vuichard et al., 2004)*

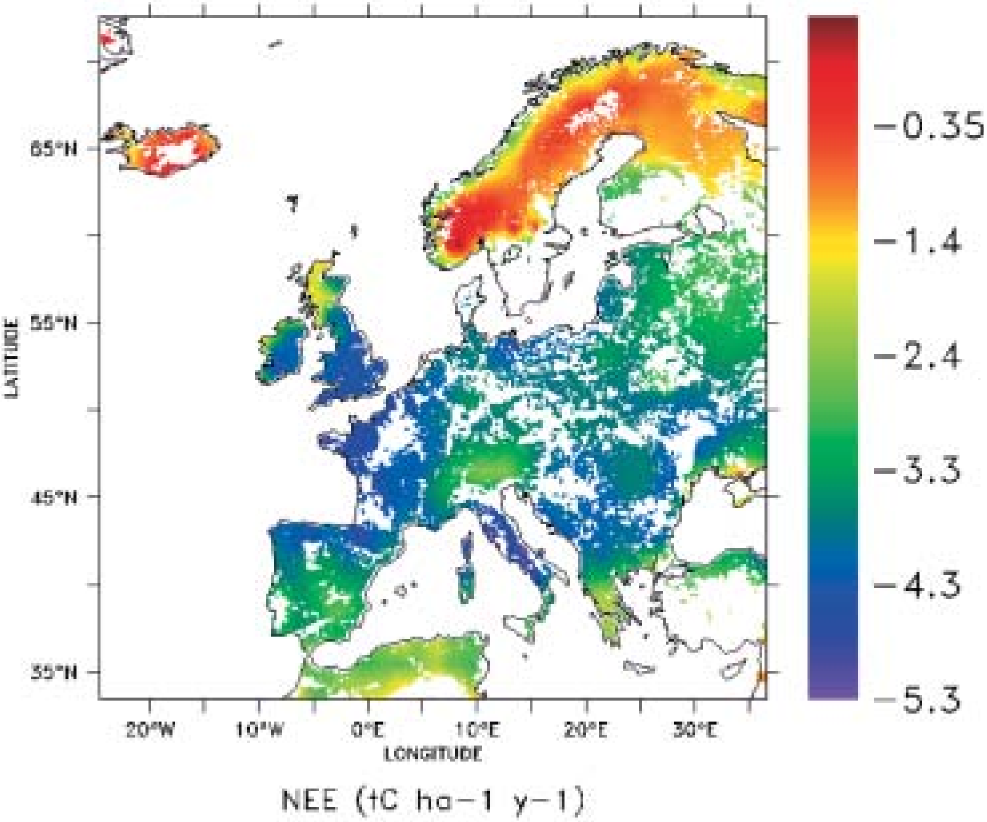


Figure 22 - *Simulated net ecosystem exchange (NEE) over Europe using the PASIM model (Vuichard et al., 2004) with N fertiliser supply allowing to reach 70% of the maximal herbage production and with a combination of grazing and cutting management, according to the forage needs for animal production during the grazing season and the stabulation periods, respectively. The stocking density at grazing is adjusted automatically by the model to reach a maximal herbage use efficiency.*

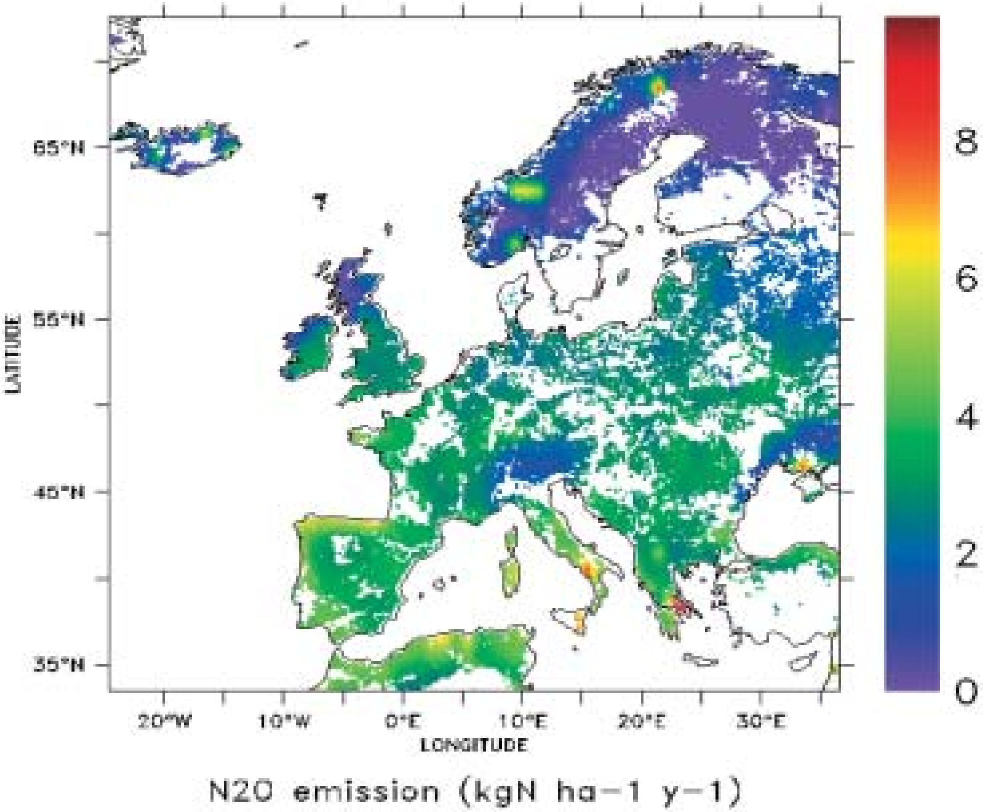


Figure 23 - *Simulated N_2O emissions over Europe using the PASIM model (Vuichard et al., 2004) with N fertiliser supply allowing to reach 70% of the maximal herbage production and with a combination of grazing and cutting management, according to the forage needs for animal production during the grazing season and the stabulation periods, respectively. The stocking density at grazing is adjusted automatically by the model to reach a maximal herbage use efficiency.*

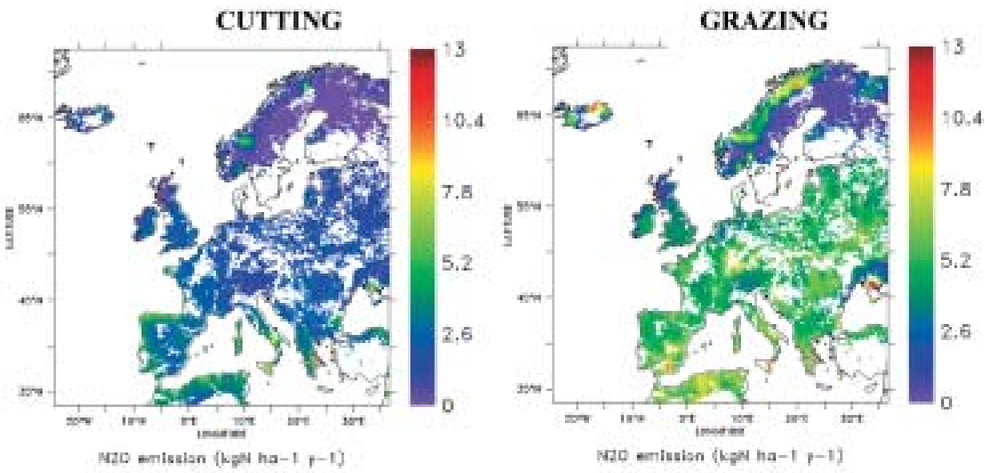


Figure 24 - *Comparison of N_2O emissions on grazed and cut grasslands using the PASIM model (Vuichard et al., 2004) with N fertiliser supply allowing to reach 70% of the maximal herbage production under cutting (A) and grazing (B) managements.*

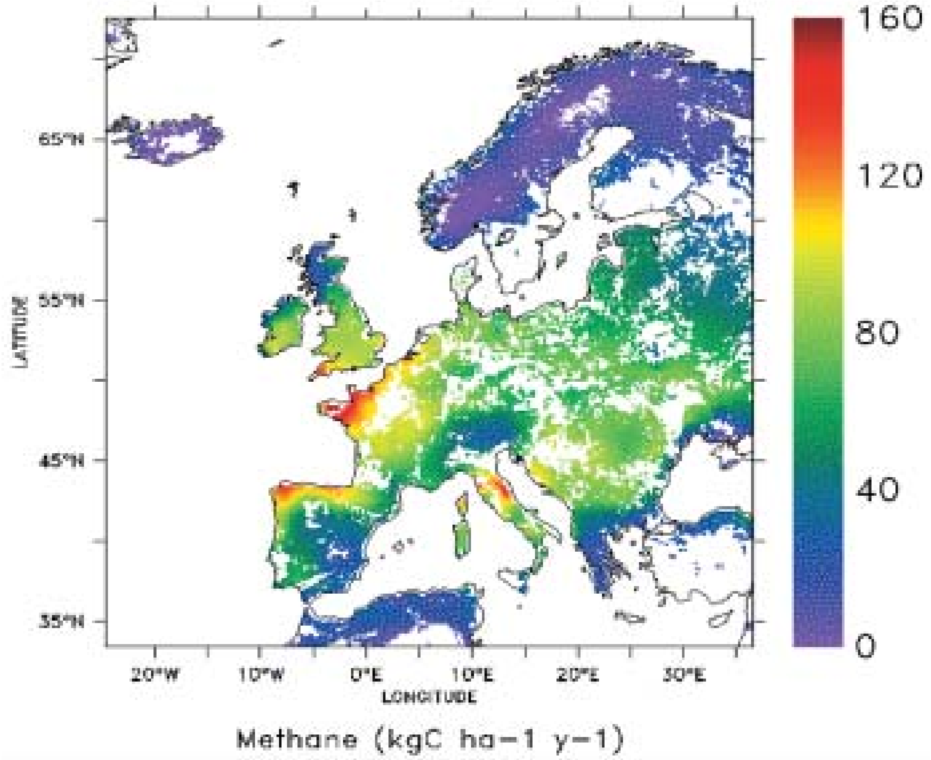


Figure 25 - *Simulated methane emissions from enteric fermentation at grazing over Europe using the PASIM model (Vuichard et al., 2004). Same run as in Figures 22 and 23.*

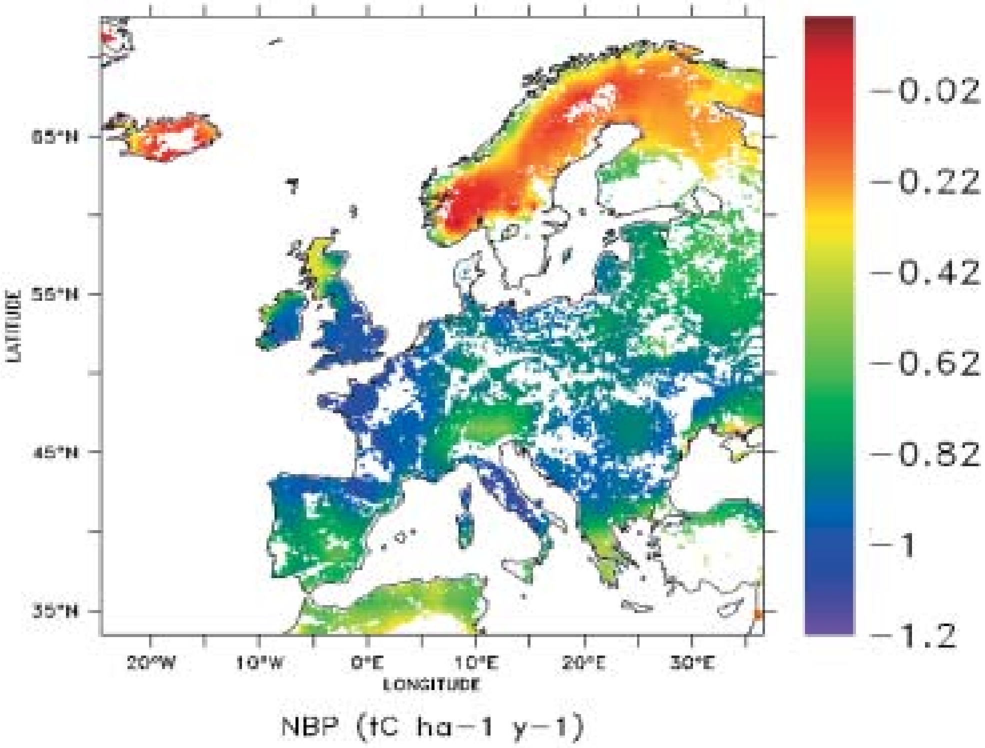


Figure 26 - *Distribution of NBP over Europe (assuming that NBP is a constant fraction of NEE) using the PASIM model (Vuichard et al., 2004). Same run as in Figures 22 and 23.*

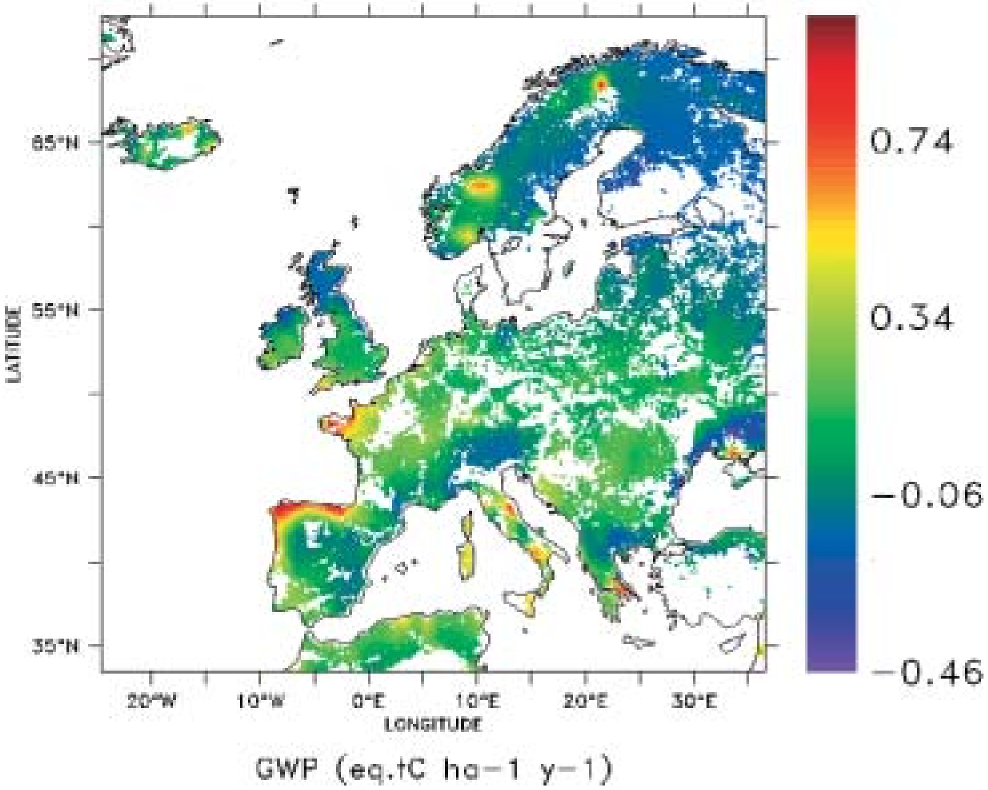


Figure 27 - *Global Warming Potential from the net exchange of GHGs with European grasslands using the PASIM model (Vuichard et al., 2004). Same run as in Figures 22 and 23.*

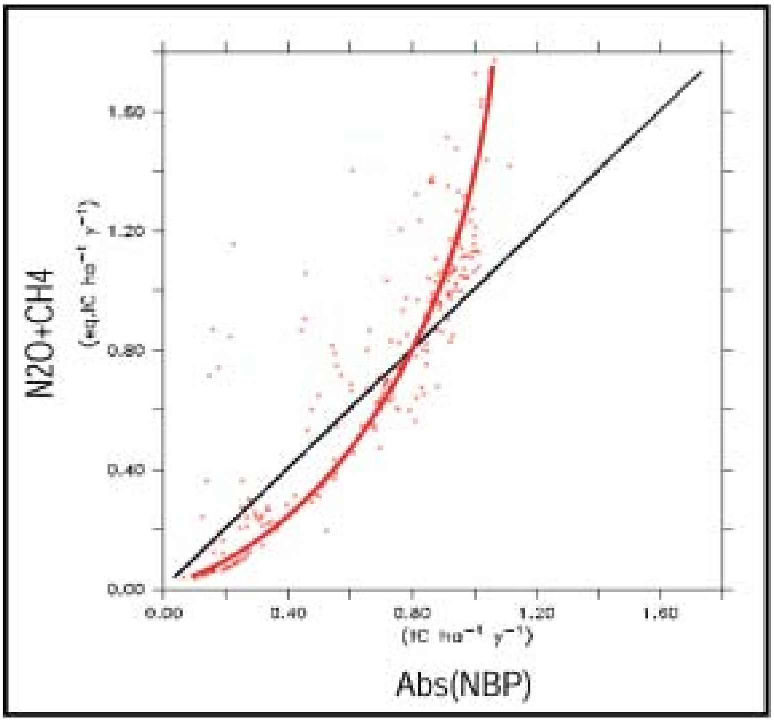


Figure 28 - *Relative contributions of methane and nitrous oxide compared to CO_2 using the PASIM model (Vuichard et al., 2004). Same run as in Figures 22 and 23.*

5.2. Land Use Change

The future of land use in Europe is unknown and will depend on the development of physical, economic and political conditions. On the other hand, changes in land use will affect human society, ecosystems and environment and there is increasing interest from research and politics in reliable information about these changes. As accurate prediction of land use change is not possible, a useful technique for the exploration of uncertain futures is the application of comprehensive, alternative scenarios. Previous attempts have developed scenarios with qualitative descriptions, short time horizons and small study regions. However, quantitative information on land use changes, that is consistent across Europe with spatial resolutions that are relevant to ecosystem scale studies, is limited.

In the European project ATEAM, it has been possible to describe an approach to develop quantitative, spatially explicit and alternative scenarios of future land use in Europe for three time slices, 2020, 2050 and 2080 (Ewert *et al*., 2004). Main emphasis is on land use for agricultural production. In a first step, driving forces were identified for different land use types (urban, agriculture, forestry and protected areas). This was done based on the Special Report on Emission Scenarios (SRES) produced by the International Panel on Climate Change (www.ipcc.ch).

SRES story lines that define demographic, economic and technological developments were interpreted for different sectors and the identified sectoral driving forces were downscaled from the global to European and regional level. The developed set of region and sector specific driving forces was then translated into quantitative, spatially explicit scenarios for different land use types. Since there is only a limited amount of earth's surface available for land use activities a land use competition hierarchy has been developed to decide about the advantage of one land use type over the others (Ewert *et al*., 2004).

Spatially explicit and alternative scenarios of future land use change are available for Europe (EU15+3) for 2020, 2050 and 2080 (Figure 29a and 29b). Land use change depends on changes in demand, productivity and/or land use protection policy (top-down approach). Technology development is the most important driver outweighing effects on land use of climate change and rising CO_2 concentration. Future agricultural land use is estimated to decrease. Decreases are substantial for A1 and A2 scenarios (economic oriented world) with about half of the agricultural land lost by 2080 mainly due to assumptions about technology development (Ewert *et al*., 2004). Decreases are least for the B1 and B2 scenario (environmental oriented world) but require protection policy and/or socio-economic measures to reduce productivity (Figure 30).

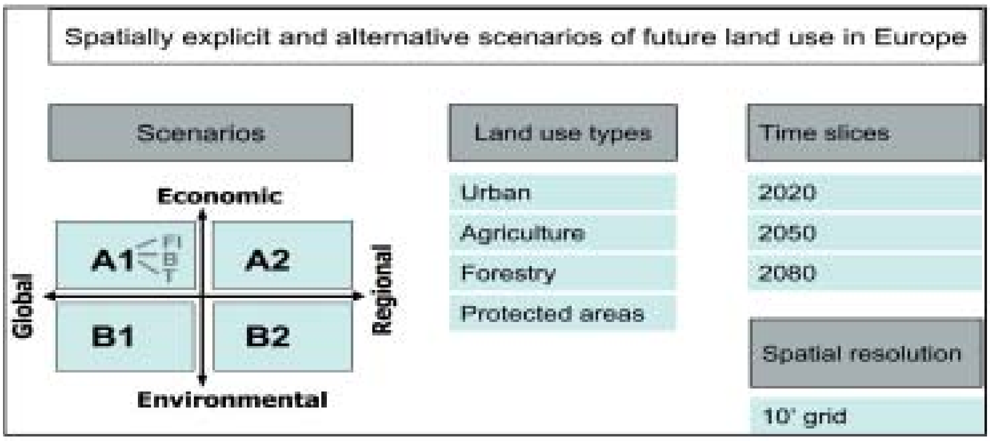


Figure 29a - *Spatial explicit and alternative scenarios of future land use in Europe*

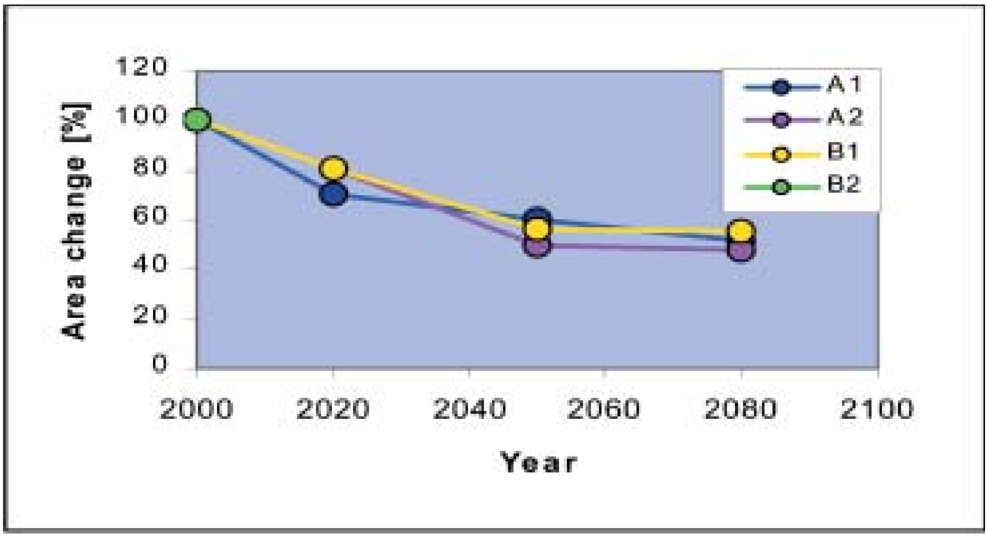


Figure 29b - *Spatial explicit and alternative scenarios of future land use in Europe*

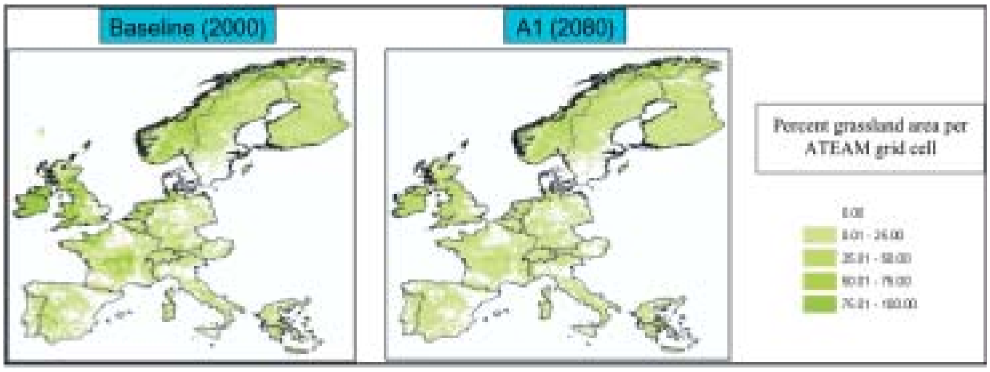


Figure 30 - *Percent grassland area per ATEAM grid cell for grassland / fodder*

According to these models, technology development is the most important driver outweighing effects on land use of climate change and rising CO_2 concentration. Decreases are substantial for A1 and A2 scenarios (economic oriented world) with about half of the grassland area lost by 2080 mainly due to assumptions about technology development. Decreases are least for the B1 and B2 scenario (environmental oriented world) but require protection policy and/or socio-economic measures to reduce productivity (Ewert *et al*., 2004)

These land use simulation results suggest that, without specific policy measures, highly intensive livestock breeding farms would predominate in Europe at the end of the century and that these farms would use only a small grassland area. The rest of the grasslands would not be used anymore for agricultural purposes and are likely to be encroached by shrubs and trees. The consequences of such possible changes in land use for carbon sequestration and for greenhouse gas emissions in Europe are unknown and clearly need further investigation.

5.3. Climate change

When climate change alone is considered, simulations of the changes in European grassland soil organic carbon (SOC) stock from 1990 to 2100 with the Roth-C model predict a 15 % decline in carbon stocks by the end of the century. However, when an increase in NPP, resulting from the rise in atmospheric CO_2 concentration is taken into account, the predicted decline in SOC stock is much smaller (Figure 31).

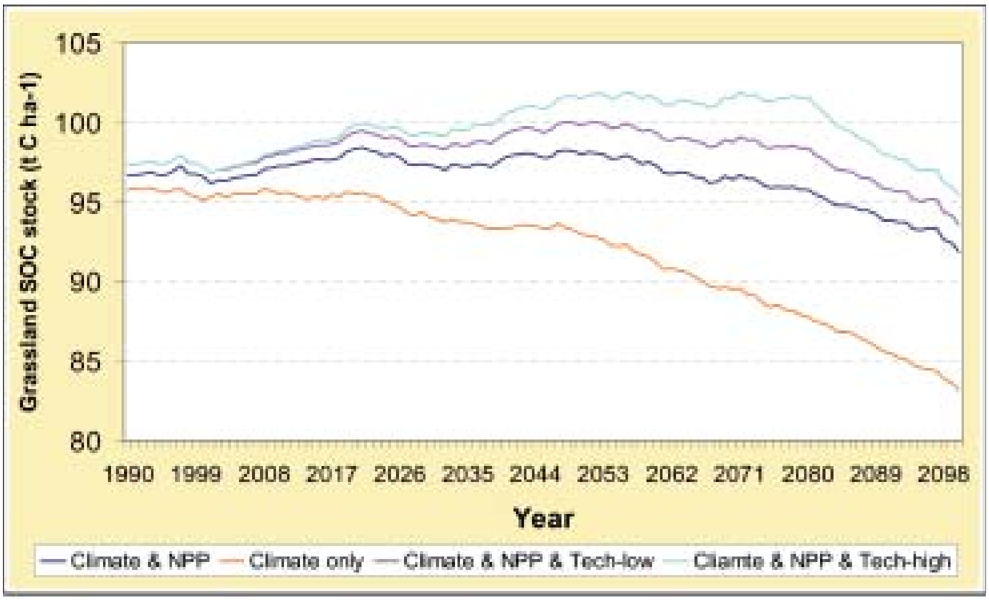


Figure 31 - *Change in mean European grassland SOC stock (excluding highly organic soils) over 1990-2100 for climate only, climate & NPP and for two scenarios where technological improvement (varieties, fertilisation, livestock management etc.) has been included: one for a higher proportion of the technology increase being transferred below ground (tech-high) than the other (tech-low). This example is for A1FI scenario as implemented by the HadCM3 model.*
After P. Smith et al. (pers. comm.)

Moreover, if herbage growth is assumed to increase through technological improvements, the soil organic carbon stock in European grasslands is predicted to remain approximately constant during the XXI$^{st.}$ century. The uncertainty associated with these predictions needs to be further evaluated, before conclusions can be drawn. Nevertheless, these simulations clearly show the interactive effects of climate change and managements practices on the soil C contents. It would also be of interest to simulate over the same time period the possible changes in the balance of non CO_2 GHG exchanged with grasslands.

• 6. Inventories, Verification & Policies

Note: Further details on GHG inventories for the European Agriculture sector are reported in the CarboEurope GHG - Specific Study Number 4: "Greenhouse Gas Emissions from European Crop" by Smith *et al* (2004).

6.1. Possibilities for improving IPCC methods to estimate GHG emissions in the future

A gap in our scientific knowledge concerns the fluxes that occur as a direct impact of a land use or management change. Instead of considering only the land use before and after a land-use or management change, more research is needed on the usual flush of (mostly) CO_2 resulting from the change itself and its duration. The transition from one usage of land to another will seriously disturb the approximate equilibrium situation which exists before the change and which will be established after some time, perhaps decades. There is scant knowledge about the causes of these enhanced fluxes, about their duration and the amounts of carbon and nitrogen lost. Long term monitoring sites with a reasonable frequency of measurements might help to assess the contribution of such a land use change to emissions.

There is scope within the current IPCC methodology to replace default emission factors and default methods with better regionally specific emission factors, where these are available, or more elaborate methodologies where these have been developed. Recent work has defined country specific values for the U.S. and found that the C sequestration rate for U.S. agricultural land was about half of the rate estimated using the default factors, primarily due to differences in the set-aside factor and the reference carbon stocks, which were computed from US data (Ogle *et al*., 2003). It would be highly desirable to perform a similar analysis to produce regionally specific values for the EU.

Another option is to develop dynamic emission factors. The IPCC default methods for calculating emission factors (see Annex) are static, i.e. they are predominantly unaffected by soil type and climate (except for CO_2) and they are assumed to be linear, i.e. they occur at a constant rate over time. However, it is known that a change in land management practice causes a non-linear change over time. Soil organic carbon, for example, is not lost at a constant rate over a 20-year period, but is better represented by an exponential loss (or gain) either as single or as multiple pools with exponential decay of soil organic carbon, which can be modelled e.g. by a first order reaction rate. It would be possible to implement emission factors based on exponential equations, or more complex models of decomposition. However, to do so, more information may be required about the soil type (such as clay content, which stabilises SOC) or climate (decomposition is sensitive to temperature and soil moisture).

Work by the GREENGRASS project has indicated that the fraction of nitrogen released from manures as nitrous oxide can be considerably higher than that for mineral fertilisers although this fraction is highly dependent upon the prevailing weather conditions. The use of zero N control plots can also give rise to difficulties in calculating emission factors. Over a period of years the proportion of N_2 fixing legumes such as clover in these plots can increase significantly giving rise to a potentially large input through biological fixation. These inputs are not adequately described in the current methodology.

Process models typically are more dynamic and capture more of the processes influencing GHG fluxes than the IPCC method which is a partial accounting method. For example, Paustian & Ogle (unpublished, data provided at Clermont-Ferrand meeting) assessed the SOC content in us agricultural soils using the IPCC method and the CENTURY simulation model (Table 20). They found that the estimated change in SOC storage for mineral soils was nearly twice as large using the century model compared to the IPCC accounting method. These results are consistent with current thought that simulation modelling will provide more complete accounting of GHG fluxes, while the IPCC method is expected to provide more conservative estimates of flux rates. However, applying simulation models requires greater detail about land use and management practices as well as considerably more resources that are not always available, and thus the IPCC method is an alternative for conducting a national inventory, given a minimal amount of information about current and past agricultural management activity.

Table 20 – Changes in *U.S. Agricultural Lands carbon contents between 1994 and 1999: comparison of two inventory methods using or not the CENTURY simulation model (TgC/yr)*

	Century Inventory	IPCC Inventory
Mineral soils	19.83	9.65
Organic soils	-9.47 (from IPCC)	-9.47
Total	10.36	0.18

These factors might be available in climatological and soil databases, but another important factor will be land-use history, which will be far more difficult to estimate. Another point to note is that soil carbon gains and losses are not symmetrical: carbon is lost more quickly when grassland is ploughed to cropland, than it is gained when croplands are reseeded to grass. This also needs to be acknowledged in any revised methodology. Any new IPCC methodology would need to consider soil types, structures and soil C contents (prior to land use changes). Similar dynamic emission factors can be envisaged also for methane and nitrous oxide. Dobbie & Smith (2003) found that annual emission factors for N_2O varied greatly from year to year, even with similar management and that several years' data were needed to produce a robust emission factor. They also recommended that differences in emission factor between various types of crop should be taken into account when compiling N_2O inventories. Also, recent study found that accounting for seasonal variations in slurry storage temperature may significantly influence accumulated emissions of CH_4 during storage, and calculated emissions for seven selected locations within Northern Europe was correlated with the mean annual temperature (Sommer *et al*., 2004).

Further, some effects of agricultural management (e.g. use of nitrification inhibitors) cannot be assessed by the IPCC method. Dynamic emission factors (which respond to climate, moisture interactions, soil type, crop type and land-use history) would provide a step closer to reality, but the quest for realism needs to be weighed against burgeoning data requirements.

Some suggestions have already been made for dynamic emission factors for nitrous oxide. Dynamic emission factors could be based upon simple (statistically derived) variables such as crop type, e.g. cereals, tuber crops, proportion of grass in the rotation, climate zone, precipitation in winter, temperature and soil type, or could be output from more complex, dynamic simulation models that include all of these interactions. Such an approach has been attempted in the USA by K. Paustian *et al.* (pers. comm. – some details in EPA, 2003). Drivers might differ according to different spatial areas, e.g. regional, continental, or national scales, or might be based on farm management types. However, data accessibility, consistency and availability need to be considered. A further consideration is how to ensure verifiability (see section 6).

Since there is scope within the current IPCC methodology to replace default emission factors and default methods it is possible to develop dynamic emission factors within the existing IPCC framework. However, with the IPCC methodologies being revised over the next 2-3 years, dynamic emission factors may appear in some sections as the new default methodologies. Research into emission factors should feed into a new IPCC emission factors database. This database will be the first step toward providing more meaningful emission factors for use in different parts of the world.

Process models (e.g. DNDC; Brown *et al*., 2002) may play a role in better determining N_2O, CO_2 and CH_4 fluxes from soils. If such models are validated first against existing (country specific) emission data, they can be used to estimate country level inventories. The advantage of such an approach is that climate and management effects can be assessed. A similar approach has already been advocated in the Joint EMEP/CORINAIR Atmospheric Emission Inventory Guidebook (EMEP/CORINAIR, 2003). Such an approach has been applied in Belgium (P. Boeckx, pers. comm.) and the UK. Since one emission may be exchanged by another, e.g. methane in saturated soil systems may be replaced by nitrous oxide if such a soil is drained, and vice versa, both experiments and models should consider nitrous oxide, methane and carbon dioxide together.

Meta-analyses of existing N_2O emission data can also be used to derive better country specific emission factors. In Belgium, statistical links between annual N_2O emissions reported in the literature and land use, seasonal climate, soil characteristics and N fertilisation rates have been established in order to provide a simple model that allows the spatial variation in environmental conditions to be taken into account in national inventories. Distinct models were developed for croplands and grassland. Emissions from croplands are sensitive to the mean temperature of the coldest month, summer precipitation and temperature, clay fraction and N fertilisation rate. Emissions from grasslands are driven by N fertilisation and summer precipitation and temperature. These empirical models are capable of explaining 60 % of the variance of annual N_2O emissions from croplands and 52 % for grasslands (Roelandt *et al*. submitted).

Upscaling of N_2O fluxes using spatial information on soil wetness and land use types may provide good inventory information (Lilly *et al*., 2003). The advantage of this approach is that areas of high emissions can be identified and application of mitigation strategies in these areas are likely to be most effective at reducing fluxes.

Finally, in order to assess the potential of mitigation options at the farm scale, there is a need to develop a specific methodology that would allow to calculate the net emissions of the three main greenhouse gases (CO_2, N_2O and CH_4) that are exchanged with the atmosphere in livestock farms.

6.2. Verification

Verification for national greenhouse gas inventories (from the IPCC, 2001b; Good Practice Guidelines) refers to ‘the activities and procedures that can be followed to establish the reliability of the data’. This usually means checking the data against empirical data or independently compiled estimates. This differs from validation, which is defined as ‘checking that the emissions and removals data has been compiled correctly in line with reporting instructions and guidelines’. If verification is interpreted strictly, estimates would be required for GHG fluxes that are independent of those used in the national report of the party to the UNFCCC. This means that for a given activity, there must be at least two independent methods for assessing the size of a GHG emission.

For cropland GHG fluxes, Smith (2004b) suggests that, if a stringent definition were used, no party would be able to meet the criteria. However, most countries would be able to meet the verification criteria by 2010 if the least stringent definition of verification were adopted (i.e. reporting of areas under a given practice [without geo-referencing] and the use of default methods and emission factors to infer a change in emissions). This approach is consistent with the Tier 1 approach of the new IPCC Good Practice Guidance on Land Use Change and Forestry (IPCC, 2004) though the Good Practice Guidance suggests that national / regional values should be used to replace defaults where they are shown to be more accurate than default values (Tier 2) or that more complex methods should be used where available (Tier 3).

For an intermediate stringency of definition (i.e. where areas under a given practice are geo-referenced [from remote sensing or ground survey], changes in carbon are derived from controlled experiments on representative climatic regions and on representative soils [or modelled using a well-evaluated, well-documented, archived model] and intensively studied benchmark sites are available for verification), only countries with the best developed inventory systems will be able to meet the requirements. Since most countries could meet verification targets if the least stringent definition of verification is used, this is the most likely to be adopted.

As in the compilation of the greenhouse gas inventory, the availability and quality of data are limiting factors for adequate verification. It is difficult in some countries even to collect reliable activity data (e.g. areas under cropland management), and much more difficult still for countries to provide data on areas under a given management practice (such as straw incorporation or zero tillage). Farm level accounting would help but would be prohibitively expensive unless collected in combination with other census data. Even if it were possible to collect data on the practices declared by a farmer / land manager, it will remain extremely difficult to verify how reliably the farmer / land-manager is implementing this practice.

Verification of mitigation measures targeting methane and nitrous oxide may be more straightforward to the extent that these involve reductions in activities, i.e., number of animals in a given category or amounts of N applied.

From the scientific perspective, the EU wishes (through projects such as CarboEurope) to obtain independent estimates of national and EU-wide GHG fluxes to verify the figures provided in national GHG inventories. At the broad level, this is possible, for example by assessing the overall C balance for Europe, but at the level of individual country inventories this may prove very difficult due to: a) the limited ability to spatially allocate emissions (and sinks) and b) the very different aim of a research project such as CarboEurope and the aims of a targeted multi-source, multi-sector verification programme. The dual constraint approach of CarboEurope (i.e. using top-down and bottom-up approaches to verify national inventories) works best with CO_2. For N_2O and for CH_4 it is possible to measure fluxes by micrometeorological methods with a network of high towers, but this measures all sources together and it is not possible to allocate measured sources to agriculture or land use. At the plot level, many of the measurements being undertaken within CarboEurope (e.g. flask measurements, eddy covariance, chamber measurements, SOC stock changes), will be very useful for verification purposes.

At the plot scale, there are two complementary methods of estimating a carbon flux, either by measurement of the CO_2 flux itself, or via measuring a change in the SOC stock (see IPCC method; section 3.3). For N_2O and CH_4, this is not possible and direct flux measurements are required. Well-documented, validated and archived dynamic simulation models may have a role to play in verification, but this raises other issues of verifiability. Other considerations include accuracy, cost and spatial variability. Smith (2004b) discusses the issue of verification (for SOC stock changes) in detail.

• 7. Conclusions: significant research needs

In this report we have summarised our current knowledge on GHG emissions from European grasslands and livestock farms, the methods to account for GHG emissions, possible GHG mitigation options in European grasslands and the constraints upon implementation of these measures. We have also provided estimates of a) the GHG fluxes from European grasslands and of b) the extent to which grassland management options can mitigate GHG emissions. We also note that the grassland CO_2 flux per unit area is highly uncertain (Janssens *et al*., 2003). We also acknowledge that the fluxes of GHGs from grasslands (especially from soils) need to be further investigated. Research priorities lie in a number of areas as detailed below.

7.1. CO_2 fluxes and soil organic C stocks in grasslands

7.1.1. Methodological issues

Given the major role of fertilisation and of disturbance by grazing and mowing in grassland ecosystems, the state variables of the vegetation (biomass, leaf area, N content) vary quickly, which in turn affects the CO_2 fluxes. It is therefore essential to record precisely both the plant and soil state variables and the management applied at grassland sites measuring CO_2 fluxes. More specifically, the animal stocking density within the footprint of the masts needs to be known, as it affects both directly (animal respiration) and indirectly (defoliation) the carbon balance.

In grazed pastures, the above-ground herbage mass varies continuously according to the balance between the growth, the death and the defoliation fluxes. When this tissue turnover reaches an equilibrium, the state variables of the vegetation tend to vary little, which makes it easier to interprete and model the role of climate factors on CO_2 fluxes. By contrast, in cut swards the rapid changes in herbage mass and in leaf area index may confound the role of climate drivers. Hence, there is a need to develop methods that will allow to better partition the variability of CO_2 fluxes between climate and management factors. Ecosystem manipulation experiments may prove to be useful as they allow to compare contrasted managements for the same climate.

In addition, more specific issues also need to be addressed:

- to which extent is the animal respiration measured correctly by the eddy flux covariance technique?
- when calculating the carbon balance of grasslands, how should we account for organic carbon exports (hay or silage, animal intake) and imports (manures, faeces and food)?
- can we reconcile the NBP values with the changes in C stocks in the soil-vegetation system?
- how can we partition the CO_2 fluxes (e.g. between gross photosynthesis, above-ground and below-ground respiration, and harvests)?

7.1.2. Drivers

Key to our understanding of soil C storage are the relationships with soil type, with temperature, with nutrients and with water availability. It is essential to consider that changes in soil carbon take place over decades and, hence, are likely to be strongly affected by the current rise in atmospheric CO_2 concentration and by climate change (temperature and rainfall). Since these factors are likely to have interactive rather than additive effects, experiments combining elevated CO_2, warming and rainfall manipulation are needed to make some progress in our ability to predict future carbon stocks in European grasslands.

More research is also required on how agricultural management affects CO_2 fluxes and C sequestration. Changes in soil carbon are difficult to quantify and monitor and changes in accumulation may take decades; however, some agricultural practices may result in rapid losses of significant amount of organic matter.

Specifically, research is required on grazing and cutting management, biological nitrogen fixation (e.g. in organic systems), extensive farming, organic soils and biomass production systems. When available, long term chronosequences are useful tools to detect changes in soil carbon stocks.

It is still debatable whether an increased N supply will increase soil C stocks (by increasing NEP) or, on the contrary, will reduce these stocks (e.g. by increasing the rate of decomposition of the soil organic matter). A simple approach which assumes an optimal balance of C and N that would maximise C storage below-ground for a given soil type would be helpful for future predictions, but it has not yet been tested sufficiently. It should be mentioned that the relation with N_2O (i.e. higher carbon stocks through improved fertilization) have a down side in terms of (continued) higher emissions of nitrous oxide.

7.2. Non-CO_2 greenhouse gases and plot scale GHG balance

All studies on grasslands should attempt to assess the combined impact of agricultural management, climate, and indirect effects (such as increasing atmospheric CO_2 concentrations and N deposition) on all biospheric GHGs (CO_2, N_2O and CH_4), not just CO_2 which has dominated previous studies. Integrative studies on C-sequestration and link with N_2O, CH_4 and NOx emissions are needed, with special emphasis on pollutants swapping. Indirect GHG emissions through nitrate leaching and ammonia emissions should also be integrated in such studies of the C and N cycles in grasslands.

Since grassland management is driven by N fertiliser additions, and a significant GHG flux from grasslands can be from nitrogenous compounds (such as nitrous oxide), a closer link between carbon and nitrogen cycling research, and the understanding of these processes (through process studies and modelling) is urgently required. For example, there is currently an insufficient understanding of the emissions of N_2O from the biological fixation of N by legumes and the net effect of N fixation on the C balance of soils. This is especially relevant for the comparison of conventional N fertilised grass vs. unfertilised grass-legume mixtures.

There are still large uncertainties concerning the methane emissions from ruminants at grazing. Further studies are required, as it appears that the herbage biomass on offer and its average chemical composition do not allow to predict accurately the emissions from enteric fermentation by grazing cattle. More experimental data are needed on organic fertilizers and on waste management systems, including anaerobic digestion. A better understanding of the relationship between manure quality (C and N pools), manure management (on and off field) and GHG emissions is required to reduce undertainties.

One additional research priority could be to develop a broader assessment that evaluates the benefits of adopting conservation management for other aspects of the environment, such as reducing nitrogen leaching, minimizing soil erosion, improving soil fertility and plant production. In addition, studies are needed to evaluate cases in which management may have unintended deleterious effects on the environment or society, even though GHG emission are reduced. These impacts will represent trade-offs that need to considered when evaluating the benefits of reducing GHG emissions through land use and management in support of public policy.

7.3. Soil process studies in grasslands

To understand some of the controls of the GHG fluxes it is essential to determine not only the fluxes but also the residence time of C and N compounds within the plant soil system. This will require (i) to identify and characterize the compartments of the soil organic matter playing a key role, (ii) to quantify some of the key internal fluxes and to monitor at the boundaries of the system fluxes towards atmosphere and hydrosphere.

We also need a better understanding of the role of different soil C pools in contributiong to C loss from soils, and in particular the importance of the process of rhizodeposition in contributing to C and N fluxes. Up to 20% of C fixed by photosynthesis can be released into the soil as a consequence of rhizodeposition during the vegetative period.

We do not have a clear understanding of the fate of this added C or the extent to which it to influences N transformations. There are opportunities through management (such as cutting and grazing), elevated CO_2 and climate change manipulation to alter rhizodeposition. However, further research is needed to quantify the impacts on net GHG emissions.

Some of the key questions for calibrating mechanistic models of soil emissions of non CO_2 trace gases relate to the soil physics (gas diffusion as affected by the macro and microporosity), as well as to the microbial controls of C and N transformations within soils. For example, interactions between urine deposition, N_2O emissions and soil organic C mineralisation are likely to occur.

7.4. Understanding the role of biodiversity for GHG balance

There is clearly a need to investigate the functional role of plant, microbial and soil fauna diversity for GHG emissions with the aim of characterising the response of the whole system to the disturbance regime induced by contrasted management systems in the long term.

Within grasslands and rangelands, the vegetation dynamics may strongly influence C storage. For example, shrub encroachment may alter the SOM dynamics as well as the balance between above and below-ground C stocks. There are also some findings which have underlined that invasive deep-rooted grasses may lead to additional C storage in tropical soils. Therefore, our understanding of the role of invasive plant species, for the C balance of grasslands needs to be reinforced.

It should also be stressed that climate change is likely to have major consequences for the geographic distribution of plant species in European grasslands. How will these changes in species composition influence the biogeochemical cycles, the C stocks and the N losses in grasslands is clearly an important issue which has not received enough attention until now.

7.5. Farm scale and data inventory

A major goal for futur research will be to reduce the large uncertainties concerning the GHG balance of European farms. Simple tools that allow to monitor the GHG balance of their farm would be extremely useful for farmers to help them achieve some GHG mitigation targets.

Guidelines for such tools at the farm scale need to be developed. Some of the important issues at this scale concern the pre-chain emissions and whether they should be included or not in the farm balance. Another aspect relates to the indirect emissions, through N leaching, N deposition and organic N exports from the farm.

Some data exist on agricultural management (and soil characteristics) at the European, national, sub-national and farm scales, but is not readily available for modelling and up-scaling. Since agricultural management is a key driver of GHG emissions, a work-programme to collect, collate and make available this data is urgently required. A meta-analysis using this data to calculate response factors for N_2O and CH_4 oxidation with respect to driving variables, e.g. climate, soil type, fertiliser type, soil organic C content, etc. is also required. Statistical approaches should be developed / improved to optimise data collection (where, when and what to measure). Well-evaluated process-based models, tested at a series of benchmark, may play a role in GHG accounting in the future. Verification of GHG emission estimates will, however, remain difficult.

Assessment of realistic mitigation and adaptation options in agriculture is needed especially at the farm scale at which the management is practiced. These need to be assessed not only for biological potential but also for economic viability, and for social, institutional and policy constraints and for potential side effects. Some R&D on knowledge transfer may also be required as successful mitigation depends heavily on this.

7.6. Development of future land-use and land management scenarios

It is widely accepted that current management practices result in significant nitrogen loss from agricultural systems with consequent economic and environmental impacts. It is anticipated that on a global basis the use of N fertiliser will strongly increase in the next 15 years. There is an urgent need to develop farming systems that utilise nitrogen inputs more efficiently and as a consequence have less detrimental environmental impacts. Opportunities for farming systems to achieve these objectives include appropriate management strategies (eg N-fertilization, species composition, grazing intensity and low-input farming).

The improved management of organic matter additions to soils is a key issue in developing more sustainable farming systems in Europe. In a review of recent data, (Six *et al.* 2004) found that the changes in carbon and nitrogen cycling are more complex than had previously been recognised. Long term benefits in terms of reductions in N_2O loss are possible, but these may depend on local site conditions. Our ability to test these hypotheses with existing datasets is very limited. A better understanding of these underlying scientific issues will be an essential basis on which to build policy instruments to regulate the environmental impact of land use changes. The outcome of this research are will be to develop new agricultural management and land-use scenarios that are appropriate for the changing demands of the XXIst Century.

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www.pik-potsdam.de/ateam

CARBOEUROPE cluster FP5 (A cluster of projects to understand and quantify the carbon balance of Europe)
Co-ordinated by Max-Planck-Institute for Biogeochemistry (Dr. Ernst-Detlef Schulze)
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CARBOEUROPE-GHG (Concerted action: Synthesis of the European Greenhouse Gas Budget)
European project n° EVK2-CT2002-20014
Co-ordinated by University of Tuscia (Prof. Dr. Riccardo Valentini)
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CARBOEUROPE-IP (FP6 Integrated Project : Assessment of the European Terrestrial Carbon Balance**)**
European project n° GOCE-CT2003-505572
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MIDAIR (Greenhouse gas mitigation for organic and conventional dairy production)
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• 9. Annex - Summary of IPCC default methods to estimate GHG emissions

The IPCC suggests default methods for estimating emissions of GHGs from agriculture and land-use change. Nitrous oxide and methane emissions are accounted for under the agricultural sector (Chapter 4 of the Revised 1996 IPCC guidelines for National Greenhouse Gas Inventories; IPCC, 1997) whereas carbon dioxide emissions are estimated in the land-use change sector (Chapter 5 of the Revised 1996 IPCC guidelines for National Greenhouse Gas Inventories; IPCC, 1997). Since field burning of agricultural residues is no longer permitted within the EU, these are not discussed here.

9.1. Nitrous oxide

The information in this section is taken directly from Chapter 4 of the Revised 1996 IPCC guidelines for National Greenhouse Gas Inventories (IPCC, 1997). Three sources of N_2O are distinguished in the IPCC methodology (IPCC, 1997): (i) direct emissions from agricultural soils, (ii) direct soil emissions from animal production (including emissions from housing to be reported under Manure Management (Section 4.2) – not discussed further in this cropland report), and (iii) N_2O emissions indirectly induced by agricultural activities.

Anthropogenic input into agricultural systems includes synthetic fertiliser, nitrogen from animal wastes, nitrogen from increased biological N-fixation, and nitrogen derived from cultivation of mineral and organic soils through enhanced organic matter mineralisation. Nitrous oxide may be produced and emitted directly in agricultural fields, animal confinements or pastoral systems or be transported from agricultural systems into ground and surface waters through surface runoff, nitrogen leaching, consumption by humans and introduction into sewage systems which transport the nitrogen ultimately into surface water. Ammonia and oxides of N (NO_x) are also emitted from agricultural systems and may be transported off-site and serve to fertilise other systems which leads to enhanced production of N_2O.

Under the IPCC methodology, agricultural systems are considered as being the same throughout the world and this methodology does not take into account different crops, soils and climates, which are known to regulate N_2O production. These factors are not considered because limited data are available to provide appropriate emission factors. The method also uses a linear extrapolation between N_2O emissions and fertiliser nitrogen application and in the indirect emissions section does not account for the probable lag time between nitrogen input and ultimate production of N_2O as a result of this nitrogen input into agricultural soils.

9.1.1. Direct nitrous oxide emissions from soils

Most studies on N_2O emissions from agricultural soils investigate the difference in N_2O production between fertilised and unfertilised fields. Emissions from unfertilised fields are considered background emissions. However, actual background emissions from agricultural soils may be higher than historic natural emissions as a result of enhanced mineralisation of soil organic matter. This is particularly observed in organic soils in both cold and warm climates over the globe. Background emissions may also be lower than historic emissions due to depletion of soil organic matter (IPCC, 1997).

According to IPCC (1997), the following sources and sinks of N_2O can be distinguished.

- Synthetic fertilisers;
- Animal excreta nitrogen used as fertiliser;
- Biological nitrogen fixation;

- Crop residue and sewage sludge application;
- Glasshouse farming (not dealt with in this report);
- Cultivation of soils with a high organic content;
- Soil sink for N_2O.

Within the IPCC methodology, all of these N_2O sources are included in the methodology, except for sewage sludge application and the soil sink for N_2O. These sources and sinks are not estimated because emissions are negligible or data are insufficient.

Synthetic fertilisers

Synthetic fertilisers are an important source of N_2O. Reviews of N_2O emissions after fertiliser addition led to an IPCC estimate of 0.0125 ± 0.01 of the applied nitrogen being directly emitted as N_2O-N. This range encompasses more than 90 per cent of the field emission values published at the time. The default emission factors for direct emissions of N_2O for Europe are:

EF_1 (fraction of N-input, kg N_2O-N/kg N) = 0.0125 (0.0025-0.0225)

EF_2 (kg N_2O-N/ha/yr) = 8 (2-15) - updated from figure of 5 in IPCC 1996 revised guidelines (IPCC, 1997) by IPCC 2001 Good Practice Guidelines (IPCC, 2001).

Animal excreta nitrogen used as fertiliser

The following is taken from IPCC (1997). Although the amount of nitrogen used as fertiliser from animal excreta is more uncertain than the amount of synthetic fertiliser used, estimates can be made, based on animal population and agricultural practices. To account for the loss of fertiliser from NH_3 volatilisation and emission of nitric oxide (NO) through nitrification after fertiliser is applied to fields, NH_3 volatilisation and NO emission factors are needed. Even though climate, soil, fertiliser placement and type, and other factors influence NH_3 volatilisation and NO_x emissions, a default emission factor of 0.1 (kg NH_3–N + NO_x–N emitted/kg N applied) can be used for synthetic fertilisers and 0.2 (kg NH_3–N + NO_x–N emitted/kg N applied) for animal waste fertiliser (0.2 is used for animal waste because of the potentially larger NH_3 volatilisation. The amount of nitrogen from these sources available for conversion to N_2O is therefore equal to 90 per cent of the synthetic fertiliser nitrogen applied and 80 per cent of the animal waste nitrogen applied.

When calculating the losses of volatile N species within manure management, N_2 losses are important. The chapter in the EMEP/CORINAIR Emission Inventory Guidebook, which is being revised with regard to these emissions in the near future, will provide a methodology. The mass flow approach, which forms the base of these calculations, can be found in Dämmgen *et al.* (2003).

Biological nitrogen fixation

Although the amount of nitrogen fixed by biological nitrogen fixation in agricultural systems can be estimated, the N_2O conversion coefficient is highly uncertain. Research indicates that biological nitrogen fixation (BNF) contributes more nitrogen for plant growth than the total amount of synthetic nitrogen fertilisers applied to crops each year. Cultivation of grain legumes, however, often results in net soil nitrogen depletion. Nitrogen from BNF may serve to fertilise an associated crop and eventually to stimulate N_2O formation. IPCC (1997) reviews studies indicating that legumes may contribute to N_2O emission in a number of ways. Atmospheric N_2 fixed by legumes can be nitrified and denitrified in the same way as fertiliser N, thus providing a source of N_2O. Additionally, symbiotically living Rhizobia in root nodules are able to denitrify and produce N_2O. Total nitrogen input is estimated by assuming that total crop biomass is about twice the mass of edible crop, and a certain nitrogen content of nitrogen fixing crop ($Frac_{NCRBF}$ – see below). A residue/crop ratio of 1 is assumed.

Crop residues

The following section is taken directly from IPCC (1997). There is only limited information concerning re-utilisation of nitrogen from crop residues and nitrogen from sewage sludge applied to agricultural lands. Although the amount of nitrogen that recycles into agricultural fields through these mechanisms may add 25-100 Tg of N/yr of additional nitrogen into agricultural soils (mainly from crop residues) the amount converted to N_2O is not known. To account for the N_2O in the inventory budget the emission factor for fertilisers is used as default and the amount of nitrogen re-entering cropped fields through crop residues is calculated from the FAO crop production data.

Nitrous oxide emissions associated with crop residue decomposition are calculated by estimating the amount of nitrogen entering soils as crop residue (FCR). The amount of nitrogen entering the crop residue pool is calculated from crop production data. Estimates of crop production (the edible part) must be roughly doubled to estimate total crop biomass. A nitrogen percentage ($Frac_{NCRBF}$ and $Frac_{NCR0}$ – see below) is then assumed to convert from kg dry biomass/yr to kg N/yr in crops. As a default N-fixing crops (pulses and soybeans) and non-N-fixing crops can be distinguished. Some of the crop residue is removed from the field as crop (approximately 45 per cent), and some may be burned (not in Europe), or fed to animals.

Cultivation of high organic content soils

Large N_2O emissions occur as a result of cultivation of organic soils (Histosols) due to enhanced mineralisation of old, N-rich organic matter. The rate of N-mineralisation is determined by the N-quality of the Histosol, management practices and climatic conditions. The range for enhanced emissions of N_2O due to cultivation is estimated to be 2-15 kg N_2O-N/ha/yr of cultivated Histosol. IPCC Good Practice Guidance (2001) adopted a default emission value of 8 kg N_2O-N ha^{-1} yr^{-1} for temperate and boreal regions.

The methodology for estimating direct N_2O emissions from agricultural fields

The *Revised IPCC 1996 Methodology* for assessing direct N_2O emissions from agricultural fields includes consideration of synthetic fertiliser (F_{SN}), nitrogen from animal waste (F_{AW}), enhanced N_2O production due to biological N-fixation (F_{BN}), nitrogen from crop residue mineralisation (F_{CR}) and soil nitrogen mineralisation due to cultivation of Histosols (F_{OS}).

In this estimate, the total direct annual N_2O-N emission is:

$$N_2O_{DIRECT} = [(F_{SN} + F_{AW} + F_{BN} + F_{CR}) \times EF_1] + F_{OS} \times EF_2 \qquad \text{(Eq. 1)}$$

where:

N_2O_{DIRECT} = direct N_2O emissions from agricultural soils in country (kg N/yr);
EF_1 = emission factor for direct soil emissions (kg N_2O-N/kg N input) ;
EF_2 = emission factor for organic soil mineralisation due to cultivation (kg N_2O-N ha/yr) ;
F_{OS} = area of cultivated organic soils within country (ha of histosols);
F_{AW} = manure nitrogen used as fertiliser in country, corrected for NH_3 and NO_x emissions and excluding manure produced during grazing (kg N/yr);
F_{BN} = N fixed by N-fixing crops in country (kg N/yr);
F_{CR} = N in crop residues returned to soils in country (kg N/yr);
F_{SN} = synthetic nitrogen applied in country (kg N/yr);
$F_{SN} = N_{FERT} \times (1\text{-}Frac_{GASF})$;
$F_{AW} = (N_{ex} \times (1\text{-}(Frac_{FUEL} + Frac_{GRAZ} + Frac_{GASM})$;
$F_{BN} = 2 \times Crop_{BF} \times Frac_{NCRBF}$;
$F_{CR} = 2 \times [Crop_0 \times Frac_{NCR0} + Crop_{BF} \times Frac_{NCRBF}\ ] \times (1\text{-}Frac_R\ ) \times (1\text{-}Frac_{BURN}\ )$; and
N_{FERT} = synthetic fertiliser use in country (kg N/yr);
$Frac_{GASF}$ = fraction of synthetic fertiliser nitrogen applied to soils that volatilises as NH_3 and NO_x (kg NH_3-N and NO_x-N/kg of N input) (see below);
N_{ex} = amount of nitrogen excreted by the livestock within a country (kg N/yr);
$Frac_{FUEL}$ = fraction of livestock nitrogen excretion contained in excrements burned for fuel (kg N/kg N totally excreted)
$Frac_{GRAZ}$ = fraction of livestock nitrogen excreted and deposited onto soil during grazing (kg N/kg N excreted) country estimate;
$Frac_{GASM}$ = fraction of livestock nitrogen excretion that volatilises as NH_3 and NO_x (kg NH_3-N and NO_x -N/kg of N excreted) (see below);
$Crop_{BF}$ = seed yield of pulses + soybeans in country (kg dry biomass/yr);
$Frac_{NCRBF}$ = fraction of nitrogen in N-fixing crop (kg N/kg of dry biomass) (see below);
$Crop_0$ = production of all other (i.e., non-N fixing) crops in country (kg dry biomass/yr);
$Frac_{NCR0}$ = fraction of nitrogen in non-N-fixing crop (kg N/kg of dry biomass) (see below);
$Frac_R$ = fraction of crop residue that is removed from the field as crop (kg N/kg crop-N) (see below);
$Frac_{BURN}$ = fraction of crop residue that is burned rather than left on field (see below).

The default values for these parameters (as given by IPCC, 1997) for Europe are as follows. $Frac_{BURN}$ = 0.10 or less (kg N/kg crop-N), $Frac_R$ = 0.45 kg N/kg crop-N, $Frac_{FUEL}$ = 0.0 kg N/kg N excreted, $Frac_{GASF}$ = 0.1 kg NH_3-N + NO_x-N/kg of synthetic fertiliser N applied, $Frac_{GASM}$ = 0.2 kg NH_3-N + NO_x-N/kg of N excreted by livestock, $Frac_{GRAZ}$ (from figures on pasture, range and paddock), $Frac_{NCRBF}$ = 0.03 kg N/kg of dry biomass, $Frac_{NCR0}$ = 0.015 kg N/kg of dry biomass.

9.1.2. Indict N_2O emissions for nitrogen used in agriculture

Pathways for synthetic fertiliser and manure input that give rise to indirect emissions considered in the Revised 1996 IPCC guidelines for National Greenhouse Gas Inventories (IPCC, 1997) are volatilisation and subsequent atmospheric deposition of NH_3 and NO_x (originating from the application of fertilisers), nitrogen leaching and runoff and human consumption of crops followed by municipal sewage treatment. Not considered are emissions from the formation of N_2O in the atmosphere from NH_3 or from food processing. Since N_2O emissions from human consumption of crops followed by municipal sewage treatment are accounted for under the waste sector, they are not discussed further here.

Atmospheric deposition of NO_x and NH_3

Atmospheric deposition of nitrogen compounds such as nitrogen oxides (NO_x) and ammonium (from NH_3) fertilise soils and surface waters and as such enhance biogenic N_2O formation. The IPCC (1997) reports rates of N_2O emissions are between 0.002 and 0.016 kg N_2O–N/kg of the amount of nitrogen deposited onto soils which is within the range of emission factors suggested for synthetic fertilisers. Emissions (EF_4) are calculated as 0.01 (0.002-0.02) kg N_2O-N /kg of NO_x-N and NH_3-N emitted annually within a country.

Although climate and fertiliser type (e.g., urea or ammonium sulphate) may influence ammonia volatilisation, the IPCC (1997) use default values for NH_3 and NO_x volatilisation of 0.1 kg nitrogen/kg synthetic fertiliser nitrogen applied to soils and 0.2 kg nitrogen/kg of nitrogen excreted by livestock ($Frac_{GASF}$ and $Frac_{GASM}$;).

Leaching and Runoff

A considerable amount of fertiliser nitrogen is lost from agricultural soils through leaching and runoff. The leached/runoff nitrogen enters groundwater, riparian areas and wetlands, rivers and eventually the coastal ocean.
Fertiliser nitrogen in ground water and surface waters enhances biogenic production of N_2O as the nitrogen undergoes nitrification and denitrification.

The fraction of the fertiliser and manure nitrogen lost to leaching and surface runoff ($Frac_{LEACH}$) may range from 0.1-0.8. A default value of 0.3 is proposed by IPCC (1997) Total nitrogen excretion is used (N_{ex}) in order to include manure produced during grazing.

$$N_{LEACH} = [N_{FERT} + N_{EX}] \times Frac_{LEACH} \qquad \text{(Eq. 2)}$$

The sum of the emission of N_2O due to N_{LEACH} in: 1) groundwater and surface drainage ($EF_{5\text{-}g}$), 2) rivers ($EF_{5\text{-}r}$), and 3) coastal marine areas ($EF_{5\text{-}e}$) is calculated to obtain the N_2O emission factor (EF_5) for NLEACH . The total amount of nitrogen eventually denitrified remains the same but some is denitrified in riparian area and wetlands before the nitrogen reaches the ocean. Default parameter values for indirect emission factors (IPCC, 1997) for Europe are as follows: $Frac_{NPR}$ = 0.16 kg N/kg of protein, $Frac_{LEACH}$ = 0.3 (0.1-0.8) kg N/kg of fertiliser or manure N.

***Groundwater**:* Assuming that all N_{LEACH} is in the form of nitrate, the IPCC recommends a default emission factor of 0.015 (EF_{5-g}; range 0.003-0.06) for N_2O from N_{LEACH} in groundwater and drainage ditches. The amount of N_2O emitted from groundwater (by upward diffusion or following entry of groundwater into surface water through rivers, irrigation, and drinking water) and agricultural drainage water is then estimated as:

$$N_2O \text{ from groundwater and agricultural drainage water} = N_{LEACH} \times EF_{5\text{-}g} \qquad \text{(Eq. 3)}$$

where EF_{5-g} = 0.015 kg N_2O-N/kg N_{LEACH} , assuming that all N_2O produced in a particular year is emitted during that year.

***Rivers**:* Once N_{LEACH} from groundwater and surface water enters rivers, additional N_2O is produced associated with nitrification and denitrification of N_{LEACH}. The IPCC (1997) method assumes that all N_{LEACH} entering rivers is nitrified once during river transport. The N_2O yield (moles N_2O-N/mol of NO_3-N) during nitrification is assumed to 0.003

for nitrification. For denitrification, a constant ratio of 0.005 for N_2O-N emission to denitrification (N_2–N production) in rivers is suggested. In summary, the emission factor for N_{LEACH} in rivers due to nitrification and denitrification [EF_{5-r}] is thus equal to 0.005 x N_{LEACH} [for nitrification] plus 0.005 x (N_{LEACH} /2) [for denitrification], or 0.0075 x N_{LEACH}. Therefore, N_2O-N produced from N_{LEACH} during river transport = N_{LEACH} x (EF_{5-r}), where EF_{5-r} = 0.0075.

Estuaries*:* Half of N_{LEACH} is assumed to be removed by denitrification in rivers in the form of N_2 and N_2O. The remaining 50 per cent of N_{LEACH} is discharged by rivers to estuaries. Nitrogen inputs to estuaries can undergo nitrification and denitrification, with associated N_2O production. For nitrification, the IPCC (1997) method assumes that half of the rivers inputs of N_{LEACH} are nitrified again in estuaries, and that the ratio of N_2O-N to NO_3–N produced is 0.005, as for rivers. For denitrification, it is assumed that 50 per cent of the N_{LEACH} that is carried to estuaries by rivers is denitrified, and the ratio of N_2O-N to denitrification (N_2-N) emitted is 0.005, as for rivers. In summary, it is assumed that 1) half of the N_{LEACH} is transported to estuaries by rivers, 2) half of the N_{LEACH} in estuaries is nitrified again in the estuary with a ratio of N_2O-N to NO_3-N of 0.005, and 3) half of the N_{LEACH} in estuaries is denitrified in the estuary with a N_2O-N to denitrification (N_2-N) ratio of 0.005.

Therefore, N_2O-N produced from N_{LEACH} in estuaries = N_{LEACH} x (EF_{5-e}) where EF_{5-e} = 0.0025. The combined emission factor [EF_5] for N_2O due to N_{LEACH} in: 1) groundwater and surface drainage (EF_{5-g} = 0.015 kg N_2O-N/kg N_{LEACH}), 2) rivers (EF_{5-r} = 0.0075 kg N_2O-N/kg N_{LEACH}), and 3) coastal marine areas (EF_{5-e} = 0.0025 kg N_2O-N/kg N_{LEACH}) is 0.025 (EF_5). Therefore:

$$\mathbf{N_{LEACH} = [N_{FERT} + N_{ex}] \times Frac_{LEACH} \text{ and } N_2O(L) = N_{LEACH} \times EF_5 \quad (Eq.\ 4)}$$

where the default values are $Frac_{LEACH}$ = 0.3 kg N/kg N input to soils and EF_5 = 0.025 kg N_2O-N/kg N_{LEACH}.

9.2. Methane

Only methane emissions from rice paddies are considered in IPCC guidelines but the new IPCC Good Practice Guidance for Land Use Change and Forestry (IPCC, 2004), considers that «the reduction of the CH_4 sink by fertilisation should be reported». Described here is the guidance from the IPCC 1996 Revised Guidelines (IPCC, 1997). Methane emissions from croplands can occur from rice fields. The area of rice grown in Europe is small and occurs mainly in southern Europe. All rice cultivated in Europe is assumed to be irrigated (IPCC, 1997). The information in this section is taken directly from Chapter 4 of the Revised 1996 IPCC guidelines for National Greenhouse Gas Inventories (IPCC, 1997). Emissions of methane from rice fields (IPCC, 1997) can be represented as follows:

$$\mathbf{Fc = EF \times A \times 10^{-12} \quad (Eq.\ 5)}$$

where:

Fc = estimated annual emission of methane from a particular rice water regime and for a given organic amendment, in Tg per year;
EF = methane emission factor integrated over cropping season, in g/m^2;

A = annual harvested area cultivated under conditions defined above. It is given by the cultivated area times the number of cropping seasons per year, i.e., in m^2/yr.

The seasonally integrated emission factor is evaluated from direct field measurements of methane fluxes for a single crop. In practice, it will be necessary to calculate the total annual emissions from a country as a sum of the emissions over a number of conditions. Total rice production can be divided into subcategories based on different biological, chemical and physical factors that control methane emissions from rice fields. In large countries, this may include different geographic regions. To account for the different conditions, F is defined as the sum of Fc (see Equation 5). This approach to emissions estimation can be represented as follows:

$$\mathbf{F = S_i\ S_j\ S_k\ EF_{ijk} \times 10^{-12}} \qquad \textbf{(Eq. 6)}$$

where:

ijk are categories under which methane emissions from rice fields may vary.

For instance, i may represent water levels in the rice fields such as fields inundated for the duration of the growing season (flooded regime) or fields under water only intermittently. This occurs either under managed irrigation when water is not readily available or when rains do not maintain flooded conditions throughout the growing season (intermittent regime).

j, k, may represent water regimes modified by other factors like organic inputs, soil textures, fertilisation regimes under each of the conditions represented by the index i, and so on. As more factors are identified, more categories need to be included. Inclusion of additional parameters should lead to an improvement of the estimate of the total emissions. The summation should include all cropping seasons.

The factors clearly identified by field experiments as being most important are (1) water regime with inorganic fertilisers (except sulphate-containing inorganic fertilisers which inhibit CH_4 production); (2) organic fertiliser applications; (3) soil type, and soil texture; (4) cultivar; and (5) agricultural practices such as direct seeding or transplanting. Data show that in continuously flooded fields, some types of organic fertilisers and certain cultivars lead to higher emissions compared to rice grown without organic amendments or intermittent or managed irrigation in which the fields are not continuously inundated and only where chemical fertilisers are used. At present there are insufficient data to incorporate most of these factors. Nonetheless, the estimates can be improved substantially by incorporating the current knowledge on water regimes, organic amendments and soil types etc. For some countries the effects of organic fertiliser can be included.

National experts are encouraged to go beyond the basic method, and add as much detail as can be scientifically justified, based on laboratory and field experiments on various amendments and theoretical calculations, to arrive at the estimate of emissions from rice cultivation in their country. These details should be incorporated into subcategories (indices j, k in Equation 6) under each of the main water management categories in Equation 5 so that they can be compared at that level with data from other countries.

For example, where emission data are available for different fertiliser types, this may be incorporated into the calculations. Each category, (e.g., continuously flooded) would be further divided as follows:

F (continuously flooded) = F (flooded/mineral fertiliser) + F (flooded/organic amendment)

This procedure would then be repeated for as many separate subcategories as have been defined. Each amendment may be incorporated in the same manner.

Scaling factors (relative to emission factors for continuously flooded rice) for irrigated rice (all European rice production) are 1.0 for continuously flooded rice, 0.5 (0.2-0.7) for intermittently flooded rice with a single aeration and 0.2 (0.1-0.3) for intermittently flooded rice with multiple aeration. The seasonally integrated methane emission factor for the only European country (Italy) represented in the Revised 1996 IPCC guidelines for National Greenhouse Gas Inventories (IPCC, 1997) is 36 (17-54) g /m^2. The arithmetic mean for all countries for which there are estimates is 20 (12-28) g /m^2 (IPCC, 1997). This value is for soils 'without organic amendments'. For conversion to methane emissions from soils 'with organic amendments', a default correction factor of 2 (range 2-5) is applied to the corresponding rice ecosystem for the 'without organic amendment' category.

9.3. Carbon dioxide

Chapter 5 (section 5.3) of the Revised 1996 IPCC guidelines for National Greenhouse Gas Inventories (IPCC, 1997) describes the methods used to calculate CO_2 emissions and uptake by soils from land-use change and management.

The principal sources/sinks of CO_2 in soils are associated with changes in the amount of organic carbon stored in soils. The IPCC methodology aims to estimate net fluxes of CO_2 due to changes in soil organic carbon stocks. CO_2 releases from liming applications are also dealt with.

The IPCC (1997) method uses a stratification of up to six major soil groups, based on major differences in their inherent carbon stocks and their response to management.

The soil groups are high clay activity mineral soils (e.g. Vertisols, Chernozems, Phaeozems, Luvisols, Vertisols, Mollisols, high-base status Alfisols), low clay activity mineral soils (e.g. Acrisols, Nitosols, Ferralsols, Ultisols, Oxisols, acidic Alfisols), sandy soils (e.g. Arenosols, sandy Regosols Psamments), volcanic soils (e.g. Andosols, Andisols), aquic (wet) soils (e.g. Gleysols Aquic suborders) and organic soils (Histosols). Of the climatic regions also used, all areas of Europe fall within the cool temperate dry, cool temperate moist, warm temperate dry or warm temperate moist zones.

The method entails calculating changes in soil organic carbon stocks due to land clearing from native vegetation (any effects of land abandonment and shifting cultivation), tillage, and carbon inputs through residue management. Organic soils are dealt with separately.

The IPCC default methodology assumes a change in carbon stocks from one equilibrium level to another over a 20-year period and calculates changes for the 0-30cm horizon only. The calculation method for mineral soils is as follows:

$$\textbf{Soil Carbon}_{managed} = \textbf{Soil Carbon}_{native} \times \textbf{Base factor} \times \textbf{Tillage factor} \times \textbf{Input factors}$$

The base factors represent changes in soil organic matter associated with conversion of the native vegetation to agricultural use, as well as setting aside cropland from production. Tillage and input factors account for effects of various management practices of lands under agricultural use. Thus these later two factors can be used to capture the changes in management trends that have occurred over the inventory period. The tillage factor accounts for changing the intensity of tillage, ranging from the most intensive practices that fully invert the soil (often referred to as conventional tillage) to the least intensive practices, such as no-till (or zero tillage). The input factor captures changes in cropping rotations, intensities or use of organic amendments that ultimately affect the carbon input to the soil due to changing overall production. Default values for soil carbon levels under native vegetation (0-30cm) are given in the Table 21.

Table 21 - *Approximate quantities of soil organic carbon under native vegetation (t C ha^{-1} to 0-30 cm depth; from IPCC, 1997) for climate zones found within Europe*

Region	High activity soil	Low activity soil	Sandy soil	Volcanic soils	Wetland soils
Cold temperate dry	50	40	10	20	70
Cold temperate moist	80	80	20	70	180
Warm temperate dry	70	60	15	70	120
Warm temperate moist	110	70	25	130	230

The coefficients used in the default calculations are shown in Table 22 below.

Table 22 - *Coefficients used in the IPCC default calculations estimating carbon stocks in mineral soils. Reproduced from IPCC (1997)* [a]

System	SG[b]	BF[c]	Tillage factor				Input factors			
			No tillage	Reduced tillage	Full tillage	Low input	Medium input	High input	Mature fallow	Short fallow
Temperate										
Long-term cultivated	A,B, C,D	0.7	1.1	1.05	1.0	0.9	1.0	1.1 / 1.2		
Long-term cultivated	E	0.6	1.1	1.05	1.0	0.9	1.0	1.1 / 1.2		
Improved pasture	All soils	1.1				ND	ND	ND		
Set aside (<20 years)	All soils	0.8				ND	ND	ND		
Set aside (>20 years)	All soils	0.9				ND	ND	ND		
Tropical										
Long-term cultivated	A,B, C,D	0.6	1.1	1.0	0.9	0.9	1.0	1.1 / 1.2		
Long-term cultivated	E	0.5	1.1	1.0	0.8	0.9	1.0	1.1 / 1.2		
Wetland (paddy) rice	All soils	1.1	ND	ND	ND	ND	ND	ND		
Shifting cultivation (including fallow)	All soils	0.8	ND	ND	ND	ND	ND	ND	1.0	0.8
Abandoned / degraded land	All soils	0.5								
Unimproved pasture	All soils	0.7				ND	ND	ND		
Improved pasture	All soils	1.1				ND	ND	ND		

<u>Notes</u>:

a Filled portions of the table, where tillage and input factors are not given, denote instances where these factors are not applicable to a management system. Where tillage or input factors were not determined (ND), information was deemed insufficient to go beyond estimating a base factor. SG = Soil Group, BF = Base Factor

b Soil groups A = High activity, B = Low activity, C = Sandy, D = Volcanic, E = Aquic

c For temperate cultivated soils, the average loss of 30% (0.7) is based on studies reported in IPCC (1997). Greater losses for cultivation of wet (aquic) soils, relative to other mineral soils, are assumed due to artificial drainage and enhanced decomposition when cultivated. Conversion to paddy rice is assumed to slightly increase carbon contents. Carbon levels in improved pastures can exceed native levels with fertilisation and species selection. Carbon under shifting cultivation (including the fallow phase) and abandoned degraded lands are based on estimates reported in IPCC (1997)

d Use of no-till is assumed to increase soil carbon by 10% over full tillage (full soil inversion) in temperate systems, based on analysis of long-term experiments in Australia, Canada, Europe and the United States; greater effects, over full tillage, are assumed for tropical systems. Reduced tillage (i.e., significant soil disturbance but without inversion) is assumed to yield small increases over full tillage

e Input factors apply to residue levels and residue management, use of cover crops, mulching, agroforestry, bare fallow frequency in semi-arid temperate systems. Low input applies to where crop residues are removed or burned, or use of bare fallow; medium input to where crop residues are retained; high input applies to where residue additions are significantly enhanced with addition of mulches, green manure, or enhanced crop residue production (1.1) or regular addition of high rates of animal manure (1.2), relative to the nominal (medium) case. Based on studies reported in IPCC (1997).

For organic soils, the method is based on the assumption that there are constant loss rates for cropland due to drainage of wetlands, and that those rates vary with climate. Losses are 0.25 t C/ha/yr in cool temperate regions, 10 t C/ha/yr in warm temperate regions, and 20 t C/ha/yr in the tropical regions. The loss rates from conversions to pasture are 25 per cent of those under cropland within each climate region. For liming, for the purposes of the inventory it is assumed that the addition rate of lime is in near equilibrium to the consumption of lime applied in previous years. Emissions associated with use of carbonate limes can thus be calculated from the amount and composition of the lime applied annually within a country.

Total annual emissions of CO_2, are calculated from i) net changes in carbon storage in mineral soil, ii) CO_2-C emissions from organic soils and iii) CO_2-C emissions from liming.