

CHAPTER VI

Mitigation the greenhouse gas balance of ruminant production systems through carbon sequestration in grasslands¹

Abstract

Soil carbon (C) sequestration (enhanced sinks) is the mechanism responsible for most of the greenhouse gas (GHG) mitigation potential in the agriculture sector. Carbon sequestration in grasslands can be determined directly by measuring changes in soil organic carbon (SOC) stocks and indirectly by measuring the net balance of C fluxes. A literature search shows that grassland C sequestration reaches on average 5 ± 30 g C/m²/year according to inventories of SOC stocks and -231 and 77 g C/m²/year for drained organic and mineral soils, respectively, according to C flux balance. Off-site C sequestration occurs whenever more manure C is produced by than returned to a grassland plot. The sum of on and off-site C sequestration reaches 129, 98 and 71 g C/m²/year for grazed, cut and mixed European grasslands on mineral soils, however with high uncertainty. A range of management practices reduce C losses and increase C sequestration: (i) avoiding soil tillage and the conversion of grasslands to arable use; (ii) moderately intensifying nutrient-poor permanent grasslands; (iii) using light grazing instead of heavy grazing; (iv) increasing the duration of grass leys; and (v) converting grass leys to grass-legume mixtures or to permanent grasslands.

With nine European sites, direct emissions of nitrous oxide (N₂O) from soil and of methane (CH₄) from enteric fermentation at grazing, expressed in carbon dioxide equivalents (CO₂eq), compensated 10 and 34 percent of

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the on-site grassland C sequestration, respectively. Digestion inside the barn of the harvested herbage leads to further emissions of CH₄ and N₂O by the production systems, which were estimated at 130 g CO₂eq/m²/year. The net balance of on- and off-site C sequestration, CH₄ and N₂O emissions reached 38 g CO₂eq/m²/year, indicating a non-significant net sink activity. This net balance was, however, negative for intensively managed cut sites indicating a source to the atmosphere. In conclusion, this review confirms that grassland C sequestration has a strong potential to partly mitigate the GHG balance of ruminant production systems. However, since soil C sequestration is both reversible and vulnerable to disturbance, biodiversity loss and climate change, CH₄ and N₂O emissions from the livestock sector need to be reduced and current SOC stocks preserved.

Key words: Climate change, CO₂, N₂O, CH₄, soil organic carbon.

IMPLICATIONS

The carbon (C) sequestration potential by grasslands and rangelands could be used to partly mitigate the greenhouse gas emissions of the livestock sector. This will require avoiding land-use changes that reduce ecosystem soil C stocks (e.g. deforestation, ploughing up long-term grasslands) and a cautious management of pastures, aiming at preserving and restoring soils and their soil organic matter (SOM) content. Combined with other mitigation measures, such as a reduction in the use of nitrogen (N) fertilizers, of fossil-fuel energy and of N rich feedstuffs by farms this may lead to substantial reductions in greenhouse gas emissions per unit land area and per unit animal product.

INTRODUCTION

Grasslands cover about one-quarter of the earth's land surface (Ojima *et al.*, 1993) and span a range of climate conditions from arid to humid. Grasslands are the natural climax vegetation in areas (e.g. the Steppes of central Asia and the prairies of North America) where the rainfall is low enough to prevent the growth of forests. In other areas, where rainfall is normally higher, grasslands do not form the climax vegetation (e.g. north-western and central Europe) and are more productive. Rangelands are characterized by low stature vegetation, owing to temperature and moisture restrictions, and found on every continent. Grasslands contribute to the livelihoods of over 800 million people including many poor smallholders (Reynolds *et al.*, 2005) and provide a variety of goods and services to support flora, fauna and human populations worldwide. On



a global scale, livestock use 3.4 billion ha of grazing land (i.e. grasslands and rangelands), in addition to animal feed produced on about a quarter of the land under crops. By 2020, this agricultural sub-sector will produce about 30 percent of the value of global agricultural output (Delgado, 2005).

Agriculture accounted for an estimated emission of 5.1 to 6.1 Gt CO₂eq/year in 2005 (10–12 percent of total global anthropogenic emissions of greenhouse gases (GHG) (IPCC, 2007) and for approximately 60 percent of nitrous oxide (N₂O) emissions and 50 percent of methane (CH₄) emissions. Between 1990 and 2005, the direct emissions of the agriculture sector have increased by 17 percent and this increase has mostly occurred in developing countries (IPCC, 2007). The GHG inventory methodology used by the Intergovernmental Panel on Climate Change (IPCC) (IPCC, 1997, 2003) only includes, however, farm emissions in the agriculture sector. Indirect GHG emissions generated by farm activity through the use of farm inputs (e.g. fertilizers, feed, pesticides) do not belong to the agriculture sector, but are covered by other sectors such as industry (e.g. for the synthesis and packaging of inorganic nitrogen (N) fertilizers and of organic pesticides) and transport (e.g. transport of fertilizers and feed). Emissions from electricity and fuel use are covered in the buildings and transport sector, respectively (IPCC, 2003).

While the sectoral approach used by IPCC is appropriate for national and regional GHG inventories, it does not reflect emissions generated directly or indirectly by marketed products. Lifecycle analyses include indirect emissions generated by farm inputs and pre-chain activities. With this approach, it was estimated that livestock production systems, from feeding import to marketed animal products, generate directly and indirectly 18 percent of global GHG emissions as measured in CO₂eq (FAO, 2006). Livestock production induces 9 percent of global anthropogenic CO₂ emissions. The largest share (i.e. 7 percent) of this derives from land-use changes – especially deforestation – caused by expansion of pastures and arable land for feed crops. Livestock production systems also emit 37 percent of anthropogenic CH₄ (see Martin, Morhavi and Doreau, 2009) most of that from enteric fermentation by ruminants. Furthermore, it induces 65 percent of anthropogenic N₂O emissions, the great majority from manure (FAO, 2006).

Agricultural ecosystems hold large C reserves (IPCC, 2001), mostly in soil organic matter (SOM). Historically, these systems have lost more than 50 Gt C (Paustian *et al.*, 1998; Lal, 1999, 2004). Agricultural lands generate very large CO₂ fluxes both to and from the atmosphere (IPCC, 2001), but the *net* flux would be small (US-EPA, 2006). Nevertheless, soil C sequestration

(enhanced sinks) is the mechanism responsible for most of the mitigation potential in the agriculture sector, with an estimated 89 percent contribution to the technical potential (IPCC, 2007) excluding, however, the potential for fossil energy substitution through non-agricultural use of biomass. Worldwide the soil organic carbon (SOC) sequestration potential is estimated to be 0.01 to 0.3 Gt C/year on 3.7 billion ha of permanent pasture (Lal, 2004). Thus SOC sequestration by the world's permanent pastures could potentially offset up to 4 percent of the global GHG emissions.

Here, we review the C sequestration potential of temperate managed grasslands, focusing on Europe, and its role for mitigating the GHG balance of livestock production systems. We address the following issues: (i) C and GHG balance of managed grasslands; (ii) vulnerability of grassland C stocks to climate change and to biodiversity loss; and (iii) the role of C sequestration for the GHG balance of ruminant production systems.

THE CARBON BALANCE OF MANAGED GRASSLANDS

Organic carbon cycling in grasslands

The nature, frequency and intensity of disturbance play a key role in the C balance of grasslands. 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 may be compensated for by organic C imports through farm manure and slurry application.

Under intensive grazing, up to 60 percent of the above-ground dry matter production is ingested by domestic herbivores (Lemaire and Chapman, 1996). However, this percentage can be much lower under extensive grazing. The largest part of the ingested C is digestible and, hence, is respired shortly after intake. The non-digestible C (25–40 percent of the intake according to the digestibility of the grazed herbage) is returned to the pasture in excreta (mainly as faeces). In most productive husbandry systems, the herbage digestibility tends to be maximized by agricultural practices such as frequent grazing and use of highly digestible forage cultivars. Consequently, in these systems the primary factor that modifies the C 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) (Soussana *et al.*, 2004). Secondary effects of grazing on the C cycle of a pasture include: (i) the role of excretal returns which, at a moderate rate of grazing intensity, could favour nutrient cycling and increase primary production, especially in nutrient-poor grasslands (De Mazancourt, Loreau and Abbadie, 1998); and (ii) the role of



defoliation intensity and frequency and of treading by animals which both reduce the leaf area and then the atmospheric CO₂ capture.

Only a small fraction of the ingested grassland C is accumulated by ruminants in meat production systems (e.g. 0.6 percent of C intake with heifers under continuous upland grazing, Allard *et al.*, 2007), but this fraction becomes much higher in intensive dairy production systems (e.g. 19–20 percent of C intake, Faverdin *et al.*, 2007). Additional C losses (approximately 3–5 percent of the digestible C) occur through CH₄ emissions from the enteric fermentation (IPCC 2006; see Martin, Morgavi and Doreau, 2009).

Processes controlling soil organic carbon accumulation

C accumulation in grassland ecosystems occurs mostly below ground. Grassland soils are typically rich in SOC, partly due to active rhizodeposition (Jones and Donnelly, 2004) and to the activity of earthworms that promote macro-aggregate formation in which micro-aggregates form that stabilize SOC for extended periods (Six *et al.*, 2002; Bossuyt, Six and Hendrix, 2005). Rhizodeposition favours C storage (Balesdent and Balabane, 1996), because direct incorporation into the soil matrix allows a high degree of physical stabilization of the SOM. Root litter transformation is also an important determinant of the C cycle in grassland ecosystems, which is affected both by the root litter quality and by the rhizosphere activity (Personeni and Loiseau, 2004 and 2005).

Below-ground C generally has slower turnover rates than above-ground C, as most of the organic C in soils (humic substances) is produced by the transformation of plant litter into more persistent organic compounds (Jones and Donnelly, 2004). Coarse SOM fractions (above 0.2 mm) have a fast turnover in soils, and the mean residence time of C in these fractions is reduced by intensive compared to extensive management (Klumpp, Soussana and Falcimagne, 2007). SOC may persist because it is bound to soil minerals and exists in forms that microbial decomposers cannot access (Baldock and Skjemstad, 2000). Therefore, SOC accumulation is often increased in clayey compared to sandy soils.

Sequestered SOC can, if undisturbed, remain in the soil for centuries. In native prairie sites in the great plains of the United States where SOC was radiocarbon (¹⁴C) dated (Follett *et al.*, 2004), mean residence time of SOC in the soil increased but its concentration decreased with depth. Nevertheless, substantial amounts of SOC remained at depth even after several millennia. In an upland grassland in France, the mean residence time of SOC also increased with depth, reaching values of 2 000–10 000 years in deep soil layer (>0.2 m)

(Fontaine *et al.*, 2007). The lack of energy supply from fresh organic matter protects ancient buried organic C from microbial decomposition (Fontaine *et al.*, 2007). Therefore, agricultural practices such as ploughing which mix soil layers and break soil aggregates accelerate SOC decomposition (Paustian *et al.*, 1998, Conant *et al.*, 2007).

While there has been a steady C accumulation in the soils of many ecosystems over millennia (Schlesinger, 1990), it is usually thought that soil C accumulation capacity is limited and that old non-disturbed soils should have reached equilibrium in terms of their C balance after several centuries (Lal, 2004). Soil C sequestration is reversible since factors such as soil disturbance, vegetation degradation, fire, erosion, nutrients shortage and water deficit may all lead to a rapid loss of SOC.

Role of land-use change for carbon sequestration

C sequestration can be determined directly by measuring changes in C pools (Conant, Paustian and Elliott, 2001) and, or, indirectly by measuring C fluxes (Table 9 and Eq. 1). SOC stocks display a high spatial variability (coefficient of variation of 50 percent, Cannell *et al.*, 1999) in grassland as compared with arable land, which limits the accuracy of direct determinations of C stock changes. The variability in SOC contents is increased by sampling to different depths (Robles and Burke, 1998; Chevallier *et al.*, 2000; Bird *et al.*, 2002) and in pastures by excretal returns concentrated in patches.

Changes in SOC through time are non-linear after a change in land use or in grassland management. A simple two-parameters exponential model has been used to estimate the magnitude of the soil C stock changes (Soussana *et al.*, 2004), showing that C is lost more rapidly than it is gained after a change in land use. Land-use change from grassland to cropland systems causes losses of SOC in temperate regions ranging from 18 percent (± 4) in dry climates to 29 percent (± 4) in moist climates. Converting cropland back to grassland uses for 20 years was found to restore 18 percent (± 7) of the native C stocks in moist climates (relative to the 29 percent loss due to long-term cultivation) and 7 percent (± 5) of native stocks in temperate dry climates (Conant, Paustian and Elliott, 2001). As a result of periodic tillage and reseedling, short duration grasslands tend to have a potential for soil C storage intermediate between crops and permanent grassland. Part of the additional C stored in the soil during the grassland phase is released when the grassland is ploughed up. The mean C storage increases in line with prolonging the lifespan of covers, i.e. less frequent ploughing (Soussana *et al.*, 2004).



Role of management for carbon sequestration in grasslands

A number of studies have analysed the effects of grassland and rangeland management on SOC stocks (Table 9). Most studies concern only the top-soil (e.g. 0–30 cm), although C sequestration or loss may also occur in deeper soil layers (Fontaine *et al.*, 2007). It is often assumed that impacts of management are greatest at the surface and decline with depth in the profile (Ogle, Conant and Paustian, 2004). A meta-analysis of 115 studies in pastures and other grazing lands worldwide (Conant, Paustian and Elliott, 2001), indicated that soil C levels increased with improved management (primarily fertilization, grazing management, and conversion from cultivation or native vegetation, improved grass species) in 74 percent of the studies considered (Table 9). Light grazing increased SOC stocks compared to enclosure and to heavy grazing (Ganjegunte *et al.*, 2005; Table 9). Some of the possible soil C sequestration opportunities for temperate grasslands in France have been calculated and compared (Table 9) for 20 year time periods (Soussana *et al.*, 2004). According to these estimates, annual C storage rates between 20 and 50 g C/m²/year are obtained for a range of options, which seem compatible with gradual changes in the forage production systems, namely: (i) reducing N fertilizer inputs in highly intensive grass leys; (ii) increasing the duration of grass leys; (iii) converting these leys to grass-legume mixtures or to permanent grasslands; and (iv) moderately intensifying nutrient-poor permanent grasslands. By contrast, the intensification of nutrient-poor grasslands developed on organic soils may lead to large C losses, and the conversion of permanent grasslands to leys of medium duration is also conducive to the release of soil C. Nevertheless, the uncertainties concerning the estimated values of C storage or release after a change in grassland management are still very high (estimated at 25 g C/m²/year).

Data from the National Soil Inventory of England and Wales obtained between 1978 and 2003 (Bellamy *et al.*, 2005) show that C was lost from most top soils across England and Wales over the survey period. Nevertheless, rotational grasslands gained C at a rate of around 10 g C/m²/year (Table 9). The Countryside Surveys of Great Britain are ongoing ecological assessments in the United Kingdom that have taken place since 1978 (Firbank *et al.*, 2003). In this survey, significant increases in soil C concentration, in the range 0.2–2.1 g/kg/year, were observed in both fertile and infertile grasslands (CLIMSOIL, 2008).

In Belgium, grasslands were reported either to be sequestering C in soils at rates of 22 or 44 g C/m²/year (Letten *et al.*, 2005a, Goidts and van Wesemael,

2007, respectively), or losing C at 90 g C/m²/year on podzolic, clayey and loam soils (Lettens *et al.*, 2005b). However, soil bulk density was estimated from pedo-transfer functions in these studies, which adds to the uncertainty since a small change in bulk density can result in a large change in stock of SOC (Smith *et al.*, 2007).

Follett and Schuman (2005) reviewed grazing land contributions to C sequestration worldwide using 19 regions. A positive relationship was found, on average, between the C sequestration rate and the animal stocking density, which is an indicator of the pasture primary productivity. Based on this relationship they estimate a 200 Mt SOC sequestration/year on 3.5 billion ha of permanent pasture worldwide. Using national grassland resource dataset and Normalized Difference Vegetation Index (NDVI) time series data, Piao *et al.* (2009) estimated that C stocks of China's grasslands increased over the past two decades by 117 and 101 g C/m²/year in the vegetation and soil compartments, respectively.

In their assessment of the European C balance, Janssens *et al.* (2003) concluded that grasslands were a highly uncertain component of the European-wide C balance in comparison with forests and croplands. They estimated a net grassland C sink of 66 ± 90 g C/m²/year over geographic Europe, but this estimate was not based on field data but on a simple model using yields and land-use data (Vleeshouwers and Verhagen, 2002).

The 2006 IPCC *Guidelines* allow for the estimation of: (i) C emissions and removals in grasslands due to changes in stocks in above and below-ground biomass; (ii) emissions of non-CO₂ GHGs (CO, CH₄, N₂O and nitrogen oxides) due to biomass burning (Fearnside, 2000); and (iii) C emissions and removals in grasslands due to changes in soil C stocks. Mineral and organic soils (peat, histosols etc) are separated for the calculations of soil C stock changes, provided that national inventory data are available for grassland soils (IPCC, 2003). Ogle *et al.* (2004) identified 49 studies dealing with effects of management practices that either degraded or improved conditions relative to nominally managed grasslands. On average, degradation reduced SOC stocks to 95 and 97 percent of C stored under nominal conditions in temperate and tropical regions, respectively. In contrast, improving grasslands with a single management activity enhanced SOC stocks by 14 and 17 percent in temperate and tropical regions, respectively, and with an additional improvement(s), stocks increased by another 11 percent. By applying these factors to managed grasslands in the United States, Ogle, Conant and Paustian (2004) found that over a 20 year period changing management could sequester up to 142 Mt C/year.



Estimating carbon sequestration from carbon flux measurements

An alternative to the direct measurement of C stock changes in grasslands is to measure the net balance of C fluxes (net carbon storage [NCS]) exchanged at the system boundaries. This approach provides a high temporal resolution and changes in C stock can be detected within one year. In contrast, direct measurements of stock change require several years or several decades to detect significant effects given the high variability among samples. The main drawback of flux measurements, however, is that several C fluxes need to be measured: (i) trace gases exchanged with the atmosphere (i.e. CO₂, CH₄, volatile organic compounds [VOC], and emissions during fires); (ii) organic C imports (manures) and exports (harvests, animal products), and (iii) dissolved C lost in waters (dissolved organic and inorganic C) and lateral transport of soil C through erosion (Figure 9). NCS (g C/m²/year) is the mass balance of these fluxes (Eq. 1):

$$\text{NCS} = (F_{\text{CO}_2} - F_{\text{CH}_4\text{-C}} - F_{\text{VOC}} - F_{\text{fire}}) + (F_{\text{manure}} - F_{\text{harvest}} - F_{\text{animal-products}}) - (F_{\text{leach}} + F_{\text{erosion}}) \quad (\text{Eq. 1})$$

where F_{CO_2} is equal to the net ecosystem exchange of CO₂ between the ecosystem and the atmosphere, which is conventionally positive for a C gain by the ecosystem. $F_{\text{CH}_4\text{-C}}$, F_{VOC} and F_{fire} are trace gas C losses from the ecosystem (g C/m²/year). F_{manure} , F_{harvest} and $F_{\text{animal-products}}$ are lateral organic C fluxes (g C/m²/year) which are either imported or exported from the system, F_{leach} and F_{erosion} are organic (and/or inorganic C losses in g C/m²/year) through leaching and erosion, respectively.

Nevertheless, depending on the system studied and its management, some of these fluxes can be neglected for NCS calculation. For instance, fire emissions by grasslands are very low in temperate regions such as Europe (i.e. below 1 g C/m²/year over 1997-2004), while they reach 10 and 100 g C/m²/year in Mediterranean and in tropical grasslands, respectively (Van der Werf *et al.*, 2006). Erosion (F_{erosion}) is also rather insignificant in permanent grasslands (e.g. in Europe), but can be increased by tillage in the case of sown grasslands. The global map of F_{erosion} created by Van Oost *et al.* (2007) indicates that grassland C erosion rates are usually below 5 g C/m²/year, even in tropical dry grasslands (Van Oost *et al.*, 2007). The total dissolved C loss by leaching was estimated by Siemens (2003) and Janssens *et al.* (2003) at 11±8 g C/m²/year for Europe. This flux tends to be highly variable depending on soil (pH, carbonate) and climate (rainfall, temperature) factors and it could reach higher values in wet tropical grasslands, especially on calcareous substrate. VOC emissions by

grassland systems are increased in the short-term by cutting and tend to be higher with legumes than with grass species (Davison *et al.*, 2008). However, these C fluxes are usually small and can easily be neglected. Therefore, with temperate managed grasslands equation 1 can be simplified as (Allard *et al.*, 2007):

$$\text{NCS} = (F_{\text{CO}_2} - F_{\text{CH}_4\text{-C}}) + (F_{\text{manure}} - F_{\text{harvest}} - F_{\text{animal-products}}) - F_{\text{leach}} \quad (\text{Eq. 2})$$

With the advancement of micrometeorological studies of the ecosystem-scale (F_{CO_2}) exchange of CO_2 (Baldocchi and Meyers, 1998), eddy flux covariance measurement techniques have been applied to grassland and rangelands. Since the measurement uses a free air technique, as opposed to enclosures, there is no disturbance of the measured area that can be freely accessed by herbivores. Ruminant's belched CO_2 (digestive + metabolic CO_2) at grazing, which can be measured by the SF_6 method (Pinares-Patino *et al.*, 2007) is included in F_{CO_2} measurements. It has no direct effect on the atmospheric CO_2 concentration, because it is 'short-cycling' C, which has been fixed by plants earlier.

Gilmanov *et al.* (2007) have analysed tower CO_2 flux measurements from 20 European grasslands covering a wide range of environmental and management conditions. F_{CO_2} varies from significant net uptake (650 g C/m²/year) to significant release (160 g C/m²/year). Four sites became CO_2 sources in some years, two of them during drought events and two of them with a significant peat horizon (Gilmanov *et al.*, 2007). Therefore, net CO_2 release ($F_{\text{CO}_2} < 0$) is associated with organic rich soils and heat stress. Indeed, a net CO_2 release was found with drained organic soils subjected to grazing in Switzerland and in New-Zealand (Rogiers *et al.*, 2008; Nieveen *et al.*, 2005) and these sites were found to lose C (i.e. negative NCS; Table 9).

Within the European (FP5 EESD) 'GreenGrass' project, the full GHG balance of nine contrasted grassland sites covering a major climatic gradient over Europe (Tables 9 and 10) was measured during two complete years (Soussana *et al.*, 2007). The sites include a wide range of management regimes (rotational grazing, continuous grazing and mowing), the three main types of managed grasslands across Europe (sown, intensive permanent and semi-natural grassland) and contrasted N fertiliser supplies. Two sites (in Ireland and in Switzerland; Table 9) were sown grass-legume mixtures, while the remainder were long-term grasslands. At all sites, the net ecosystem exchange (NEE) of CO_2 was assessed using the eddy covariance technique.



CH₄ emissions resulting from enteric fermentation of the grazing cattle were measured *in situ* at four sites using the sulfur hexafluoride (SF₆) tracer method. N₂O emissions were monitored using various techniques (GC-cuvette systems, automated chambers and tunable diode laser).

The average C storage was initially estimated at 104 ± 73 g C/m²/year, but without accounting for C leaching and for C exports in animal products (Soussana *et al.*, 2007). NCS and component fluxes are shown in Figure 10. Results, corrected for animal exports ($F_{\text{animal-products}}$) and for C leaching (F_{leach}), show that NCS varied between 50 and 129 g C/m²/year and was higher in grazed than in cut grasslands (Figure 10). Across sites, NCS declined with the degree of herbage utilization by herbivores through grazing and cutting (Soussana *et al.*, 2007), which underlines that grassland C sequestration per unit area is favoured by extensive management provided that nutrients are not limiting (Klumpp, Soussana and Falcimagne, 2007; Allard *et al.*, 2007). The uncertainty associated with NCS can be estimated using Gaussian error propagation rules and accounting for site number in each management type. NCS uncertainty reached 25 and 80 percent of the mean (data not shown) for grazed and for cut and mixed systems, respectively. Indeed, Ammann *et al.* (2007) reported that cutting and manure application introduce further uncertainties in NCS estimates.

A literature search shows that grassland C sequestration reaches on average 5 ± 30 g C/m²/year according to inventories of SOC stocks and 22 ± 56 g C/m²/year according to C flux balance (Table 9). These two estimates are therefore not significantly different, although there has not yet been any direct comparison at the same site between C flux and C stock change measurements. According to both flux (-231 and 77 g C/m²/year, respectively, Table 9A) and inventory (Bellamy *et al.*, 2005) methods, organic soils would be more susceptible to losing carbon than mineral soils, which underlines the need to preserve high soil C stocks.

Carbon flux studies show that NCS is affected by a number of site-specific factors, including grassland type (newly established vs.-permanent, [Byrne *et al.*, 2005]), N fertilizer supply (Ammann *et al.*, 2007), grazing pressure (Allard *et al.*, 2007), drainage (Rogiers *et al.*, 2008, Nieveen *et al.*, 2005) and burning (Suyker and Verma, 2001) (Table 9). In addition, annual rainfall, temperature and radiation (Hunt *et al.*, 2004, Ciais *et al.*, 2005, Gilmanov *et al.*, 2007, Soussana *et al.*, 2007) play an important role for the variability in NCS between years and between sites. Other possibly overlooked factors in C flux studies include past changes in land use (e.g. from arable to grassland)

and grassland management (e.g. increased fertilization, reduced herbage utilization by grazing and cutting) which have carry-over effects on soil C pools. In addition, the recent rise in air temperature, in atmospheric CO₂ concentration and in N deposition has enhanced plant growth in northern mid-latitudes and high latitudes (Nemani *et al.*, 2003). Global change would therefore force grassland soils out of equilibrium, possibly leading to a transient increase in SOC stocks in temperate regions as a result of increased net primary productivity. Further research is needed to disentangle such global factors from management factors, in order to attribute grassland C sequestration to direct anthropogenic changes (land use and land management) and/or to climatic and atmospheric changes.

THE GREENHOUSE GAS BALANCE OF MANAGED GRASSLANDS

When assessing the impact of land use and land-use change on GHG emissions, it is important to consider the impacts on all GHG (Robertson, Paul and Harwood, 2000). N₂O and CH₄ emissions are often expressed in terms of CO₂eq, which is possible because the radiative forcing of N₂O, CH₄ and CO₂, can be integrated over different timescales and compared with that for CO₂. For example, over the 100 year timescale, on a kilogram for kilogram basis, one unit of N₂O oxide has the same global warming potential as 298 units of CO₂ (GWP_{N₂O}=298), whereas one unit of CH₄ has the same GWP as 25 units of CO₂ (GWP_{CH₄}=25) (IPCC, 2007). An integrated approach is needed to quantify in CO₂eq the fluxes of all three trace gases (CO₂, CH₄, N₂O).

Management choices to reduce emissions involve important trade-offs: for example, preserving grasslands and adapting their management to improve C sequestration in the soil may actually increase N₂O and CH₄ emissions at farm scale. Since agricultural management is one of the key drivers of the sequestration and emission processes, for grasslands there is potential to reduce the net GHG (NGHG flux, expressed in CO₂eq. CH₄ emissions by enteric fermentation under grazing conditions are reviewed in detail by Martin, Morgavi and Doreau (2009). Below, we focus on N₂O emissions and on the GHG balance in CO₂eq.

Nitrous oxide emissions from grassland soils

Biogenic emissions of N₂O from soils result primarily from the microbial processes nitrification and denitrification. N₂O is a by-product of nitrification and an intermediate during denitrification. Nitrification is the aerobic



microbial oxidation of ammonium to nitrate and denitrification is the anaerobic microbial reduction of nitrate through nitrite, nitric oxide and N_2O to N_2 (nitrogen gas). N_2O is a gaseous product that may be released from both processes to the soil atmosphere.

Major environmental regulators of these processes are temperature, pH, soil moisture (i.e. oxygen availability) and C availability (Velthof and Oenema, 1997). In most agricultural soils, biogenic formation of N_2O is enhanced by an increase in available mineral N, which in turn increases nitrification and denitrification rates. Hence, in general, addition of fertilizer N or manures and wastes containing inorganic or readily mineralizable 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 (Skiba and Smith, 2000).

For given soil and climate conditions, N_2O emissions are likely to scale with the nitrogen fertilizer inputs. Therefore, the current IPCC (2003) methodology assumes a default emission factor (EF_1) of 1 percent (range 0.3 to 3 percent) for non-tropical soils emitted as N_2O per unit N input N (0.003 – 0.03 kg N_2O -N/kg N input).

N_2O emissions in soils usually occur in “hot spots” associated with urine spots and particles of residues and fertilizers, despite the diffuse spreading of fertilizers and manure (Flechard *et al.*, 2007). N_2O emissions from grasslands also tend to occur in short-lived bursts following the application of fertilizers (Leahy, Kiely and Scanlon, 2004; Clayton *et al.*, 1997). Temporal and spatial variations contribute large sources of uncertainty in N_2O fluxes at the field and annual scales (Flechard *et al.*, 2005). The overall uncertainty in annual flux estimates derived from chamber measurements may be as high as 50 percent owing to the temporal and spatial variability in fluxes, which warrants the future use of continuous measurements, if possible at the field scale (Flechard *et al.*, 2007). In the same study, annual emission factors for fertilized systems were highly variable, but the mean emission factor (0.75 percent) was substantially lower than the IPCC default value of 1.0 percent for direct emissions of N_2O from N fertilizers (Flechard *et al.*, 2007).

The relationship, on a global basis, between the amount of N fixed by chemical, biological or atmospheric processes entering the terrestrial biosphere, and the total emission of N_2O shows an overall conversion factor of 3–5 percent (Crutzen *et al.*, 2007). This factor is covered only in part by the 1 percent of direct emissions factor. Additional indirect emissions, resulting

from further N₂O emissions at the landscape scale, are also accounted for by IPCC (2003).

Methane exchanged with grassland soils

In soils, CH₄ 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. CH₄ emissions from freely drained grassland soils are, therefore, negligible. In fact, aerobic grassland soils tend to oxidise CH₄ at a higher rate than cropland soil (6 and 3 kg CH₄/ha/year respectively), but less so than uncultivated soils (Boeckx and Van Cleemput, 2001). In contrast, in wet grasslands as in wetlands, the development of anaerobic conditions in soils may lead to CH₄ emissions. In an abandoned peat meadow, CH₄ emissions were lower in water unsaturated compared with water saturated soil conditions (Hendriks *et al.*, 2007). Keppler *et al.* (2006) have shown the emissions of low amounts of CH₄ by terrestrial plants under aerobic conditions. However, this claim has not since been confirmed and was shown to be caused by an experimental artefact (Dueck *et al.*, 2007).

Budgeting the greenhouse gas balance of grasslands

Budgeting equations can be extended to include fluxes (F_{CH₄-C} and F_{N₂O}) of non-CO₂ radioactively active trace gases and calculate a net exchange rate in CO₂eq (NGHG, g CO₂/m²/year, Eq. 3), using the global warming potential (GWP) of each gas at the 100 year time horizon (IPCC, 2007):

$$\text{NGHG} = k_{\text{CO}_2}(\text{NCS} + F_{\text{CH}_4\text{-C}}) - \text{GWP}_{\text{CH}_4} F_{\text{CH}_4} - \text{GWP}_{\text{N}_2\text{O}} F_{\text{N}_2\text{O}} \quad (\text{Eq. 3})$$

where $k_{\text{CO}_2} = 44/12 \text{ g CO}_2 \text{ g C}$, F_{CH_4} is the methane emission (g CH₄/m²/year) and $F_{\text{N}_2\text{O}}$ is the N₂O emission (g N₂O/m²/year). CH₄ is not double counted as CO₂ in equation 4, since F_{CH₄-C} is added to NCS.

On average of the nine sites covered by the 'GreenGrass' European project, the grassland plots displayed annual N₂O and CH₄ emissions of 39 and 101 g CO₂eq/m²/year, respectively (Table 10). Hence, when expressed in CO₂eq, emissions of N₂O and CH₄ compensated 10 and 34 percent of the onsite grassland C sequestration, respectively. The mean on-site NGHGH reached 198 g CO₂eq/m²/year, indicating a sink for the atmosphere. Nevertheless, sites that were intensively managed by grazing and cutting had a negative NGHGH and were therefore estimated to be GHG sources in CO₂eq.



VULNERABILITY OF SOIL ORGANIC CARBON TO CLIMATE CHANGE

Although the ancient C located in the deep soil is presumably protected from microbial decomposition by a lack of easily degradable substrates (Fontaine, Mariotti and Abbadie, 2003), soil C stocks in grassland ecosystems are vulnerable to climate change. The 2003 heat-wave and drought reduced total gross primary productivity over Europe by 30 percent, which resulted in a strong anomalous net source of CO₂ (0.5 Gt C/year) to the atmosphere and reversed the effect of four years of net ecosystem C sequestration (Ciais *et al.*, 2005). An increase in future drought events could therefore turn temperate grasslands into C sources, contributing to positive C-climate feedbacks already anticipated in the tropics and at high latitudes (Betts *et al.*, 2004; Ciais *et al.*, 2005; Bony *et al.*, 2006). Gilmanov *et al.* (2005) have also shown that a source type of activity is not an exception for the mixed prairie ecosystems in North America, especially during years with lower than normal precipitation.

The atmospheric conditions that result in such heat wave conditions are likely to increase in frequency (Meehl and Tebaldi, 2004) and may approach the norm by 2080 under scenarios with high GHG emissions (Beniston, 2004; Schär and Jendritzky, 2004). The rise in atmospheric CO₂ reduces the sensitivity of grassland ecosystems to drought (Morgan *et al.*, 2004) and increases grassland productivity by 5–15 percent depending on water and nutrients availability (Soussana, Casella and Loiseau, 1996; Soussana and Hartwig, 1996; Tubiello *et al.*, 2007). However, these positive effects are unlikely to offset the negative impacts of high temperature changes and reduced summer rainfall, which would lead to more frequent and more intense droughts (Lehner *et al.*, 2006) and, presumably, C loss from soils. The possible implication of climate change was studied by Smith *et al.* (2005) who calculated soil C change using the Rothamsted C model and using climate data from four global climate models implementing four IPCC emission scenarios (SRES). Changes in net primary production (NPP) were calculated by the Lund–Potsdam–Jena model. Land-use change scenarios were used to project changes in cropland and grassland areas. Projections for 1990–2080 for mineral soil show that climate effects (soil temperature and moisture) will tend to speed decomposition and cause soil C stocks to decrease, whereas increases in C input because of increasing NPP could slow the loss.

According to empirical niche-based models, projected changes in temperature and precipitation are likely to lead to large shifts in the distribution of plant species, with negative effects on biodiversity at regional

and global scales (Thomas *et al.*, 2004; Thuiller *et al.*, 2005). Although such model predictions are highly uncertain, experiments do support the concept of fast changes in plant species composition and diversity under elevated CO₂, with complex interactions with warming and changes in rainfall (Teyssonneyre *et al.*, 2002; Picon-Cochard *et al.*, 2004). Indeed, warming and altered precipitation have been shown to affect plant community structure and species diversity in rainfall manipulation experiments (Zavaleta *et al.*, 2003; Klein, Harte and Zhao *et al.*, 2005).

Biodiversity experiments have shown causal relationships between species number or functional diversity, ecosystem productivity (e.g. Tilman, Lehman and Thomson, 1997; Hector *et al.*, 1999; Röscher *et al.*, 2005) and C sequestration (Tilman, Reich and Knops, 2006a and 2006b, Klumpp and Soussana, 2009). Therefore, another threat to C sequestration by grassland soils stems from the rapid loss of plant diversity that is projected under climate change.

THE ROLE OF GRASSLAND C SEQUESTRATION FOR THE GHG BALANCE OF LIVESTOCK SYSTEMS

There are still substantial uncertainties in most components of the GHG balance of livestock production systems. Methods developed for national and global GHG inventories are inaccurate at the farm scale. Livestock production systems can be ranked differently depending on the approach (plot scale, on farm budget, lifecycle analysis) and on the criteria (emissions per unit land area or per unit animal product) selected (Schils *et al.*, 2007). Moreover, C sequestration (or loss) plays an important (Table 9D), but often neglected, role in the farm GHG budget.

C transfer between different fields is very common in livestock production systems. The application of organic manure to certain fields may also vary strongly from year to year (depending for example on the nutrient status). To date, grassland C sequestration has mostly been studied at the field scale, neglecting the post-harvest fate of the cut herbage. The calculation of NCS considers that the total C in the harvested herbage returns to the atmosphere within one year. This is usually not the case, since the non-digestible C in this pool will be excreted by ruminants and incorporated into manure that will be spread after storage either on the same or on another field. Off-site C sequestration will occur whenever more C manure is produced by than returned to a grassland plot. To make some progress, we estimate below the off-site C and GHG balance of the harvested herbage.



Carbon balance during housing

When considering an off-site balance, the system boundaries need to be defined. In the barn, ruminant's digestion of the harvested herbage (F_{harvest}) leads to additional C losses as respiratory CO_2 and CH_4 from enteric fermentation and to the production of animal effluents (manure). The manure generated by harvests from a given grassland field will be brought to other fields (grassland or arable) thereby contributing to their own C budgets. To avoid double counting, we only attribute to a given grassland field the surplus, if any, of decomposed C manure that it generates compared with the amount of manure it receives (Figure 11). On-site decomposition of manure C supplied to the studied grassland field contributes to ecosystem respiration (F_{CO_2} , Eq. 2) and is therefore not double counted as an off-site CO_2 flux.

Off-site C sequestration ($\text{NCS}_{\text{@barn}}$) is calculated as the SOC derived from cut herbage manure that is not returned to the grassland, taking into account CH_4 emission from manure management. By adding off-site C sequestration and on-site C sequestration (NCS), an attributed NCS (Att-NCS) is calculated as:

$$\text{Att-NCS} = \text{NCS} + \text{NCS}_{\text{@barn}} = \text{NCS} + f_{\text{humif}} \cdot \text{Max}[0, (1 - f_{\text{digest}})F_{\text{harvest}} - F_{\text{manure}}] - F_{\text{CH4manure_C}} \quad (\text{Eq. 4})$$

where: f_{humif} is the fraction of non labile C in manure; f_{digest} is the proportion of ingested C that is digestible and $F_{\text{CH4manure_C}}$ is CH_4 emission from farm effluents calculated according to the IPCC (2006) Tier 2 method in CO_2 -Ceq (Figure 11). The fraction of non-labile C in manure (f_{humif}) varies between 0.25 and 0.45 (Soussana *et al.*, 2004).

Equation 3 assumes that: (i) all harvested C is ingested by ruminants (no post-harvest losses); and (ii) that the non-digestible fraction returned as excreta is used for spreading. These assumptions could lead to an overestimation of the attributed NCS, since additional C losses take place after forage harvests (during hay drying and silage fermentation) as well as in manure storage systems. However, these losses concern the degradable fraction of manures and are thus already accounted for by the f_{humif} coefficient.

$\text{NCS}_{\text{@barn}}$ reached 21.5 and 27 g C/m²/year for mixed and cut systems, respectively. Therefore, Att-NCS, which includes $\text{NCS}_{\text{@barn}}$, was higher in grazed (129 g C/m²/year) than in cut and mixed grassland systems (98.5 and 71 g C/m²/year, respectively). These estimates do not include C emissions from machinery, which are higher in cut (e.g mowing, silage making)

compared with grazed systems, but are not part of the AFOLU sector and are not discussed in this review.

Greenhouse Gas balance during housing

GHG emissions from manure management include direct emissions of CH₄ and N₂O, as well as indirect emissions of N₂O derived from ammonia/nitrogen oxides. Quantification of GHG emissions from manure are typically based on national statistics for manure production and housing systems combined with emission factors that have been defined by the IPCC or nationally (Petersen, Olesen and Heidmann, 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 CH₄ conversion factors and N₂O emission factors, respectively. However, even within each category the variations in manure composition and storage conditions can lead to highly variable emissions in practice. 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 CH₄ and N₂O emissions from manure (Sommer, Petersen and Moller, 2004).

The fraction of non-labile C (f_{humit}) in manure increases from 0.25 to about 0.5 after composting (Rémy and Marin-La Flèche, 1976). During composting, the more degradable organic compounds are decomposed and the residual compounds, which tend to have a longer life span, increase in concentration. In one study, cumulative C losses during storage and after incubation in the soil accounted for 60 and 54 percent of C initially present in composted and anaerobically stored manure, respectively (Thomsen and Olesen, 2000).

In order to account for: (i) the offsite CO₂, CH₄ and N₂O emissions resulting directly from the digestion by cattle of the forage harvests; (ii) from their contribution to CH₄ and N₂O emissions by farm effluents; and (iii) the manure and slurry applications which add organic C to the soil, an NGHGAtt-NGHG balance was adapted from Soussana *et al.* (2007) as:

$$\text{Att-NGHG} = k_{\text{CO}_2}(\text{Att-NCS} + F_{\text{CH}_4\text{-C}}) - \text{GWP}_{\text{CH}_4}(F_{\text{CH}_4} + F_{\text{CH}_4\text{@barn}} + F_{\text{CH}_4\text{-manure}}) - \text{GWP}_{\text{N}_2\text{O}}(F_{\text{N}_2\text{O}} + F_{\text{N}_2\text{O-manure}}) \quad (\text{Eq. 5})$$



where $F_{\text{CH}_4@barn}$ is CH_4 emission by enteric fermentation at barn ($\text{g CH}_4/\text{m}^2/\text{year}$), $F_{\text{CH}_4\text{-manure}}$ ($\text{g CH}_4/\text{m}^2/\text{year}$) and $F_{\text{N}_2\text{O-manure}}$ ($\text{g N}_2\text{O}/\text{m}^2/\text{year}$) are the CH_4 and direct N_2O emissions from farm effluents, respectively, which were calculated according to the IPCC (2003) Tier 2 method (Table 10).

Estimated CH_4 emissions at barn from cut herbage reached up to $447 \text{ g CO}_2\text{eq}/\text{m}^2/\text{year}$ (Table 10) and were therefore an important component of the attributed NGHG budget of the cut sites. The attributed GHG balance was positive for grazed sites (indicating a sink activity), but was negative for cut and mixed sites (indicating a source activity) (Table 10). Therefore, grazing management seems to be a better strategy for removing GHG from the atmosphere than cutting management. However, given that the studied sites differed in many respects (climate, soil and vegetation) (Soussana *et al.*, 2007), this hypothesis needs to be further tested.

Taken together, these results show that managed grasslands have a potential to remove GHG from the atmosphere, but that the utilization by ruminants of the cut herbage may lead to large non- CO_2 GHG emissions in farm buildings which may compensate this sink activity. Data from a larger number of flux sites and from long-term experiments will be required to upscale these results at regional scale and calculate GHG balance for a range of production systems. In order to further reduce uncertainties, C and N fluxes are investigated for a number of additional grassland and wetland sites (e.g. CarboEurope and NitroEurope research project). Grassland ecosystem simulation models have also been used for upscaling these fluxes (Levy *et al.*, 2007; Vuichard *et al.*, 2007a) in order to estimate the C and GHG balance at the scale of Europe. Two main problems were identified: (i) the lack of consistent grassland management data across Europe; and (ii) the lack of detailed grassland soil C inventories for soil model initialization (Vuichard *et al.*, 2007a).

Including carbon sequestration in greenhouse gas budgets at farm scale

A grazing livestock farm consists of a productive unit that converts various resources into outputs as milk, meat and sometimes grains. In Europe, many ruminant farms have mixed farming systems: they produce the roughage themselves and, most often, part of the animal feed and even straw that is eventually needed for bedding. Conversely, these farms recycle animal manure by field application. Most farms purchase some inputs, such as fertilizers and feed, and they always use direct energy derived from fossil

fuels. The net emissions of GHG (CH_4 , N_2O and CO_2) are related to C and N flows and to environmental conditions.

To date, only few recent models have been developed to estimate the farm GHG balance (Schils *et al.*, 2007). Most models have used fixed emission factors both for indoor and outdoor emissions (e.g. FARM GHG, [Olesen *et al.*, 2006, Lovett *et al.*, 2006]). Although these models have considered the on and off-farm CO_2 emissions (e.g. from fossil fuel combustion), they did not include possible changes in soil C resulting from the farm management. Moreover, as static factors are used rather than dynamic simulations, the environmental dependency of the GHG fluxes is not captured by these models.

A dynamic farm-scale model (FarmSim) has been coupled to mechanistic simulation models of grasslands (PASIM, [Riedo *et al.*, 1998; Vuichard *et al.*, 2007b]) and croplands (CERES ECC). In this way, C sequestration by grasslands can be simulated (Soussana *et al.*, 2004) and included in the farm budget. The IPCC methodology Tier 1 and Tier 2 is used to calculate the CH_4 and N_2O emissions from cattle housing and waste management systems. The NGHGBalance at the farm gate is calculated in CO_2eq . Emissions induced by the production and transport of farm inputs (fuel, electricity, N-fertilizers and feedstuffs) are calculated using a full accounting scheme based on life cycle analysis. The FarmSim model has been applied to seven contrasted cattle farms in Europe (Salètes *et al.*, 2004). The balance of the farm gate GHG fluxes leads to a sink activity for four out of the seven farms. When including pre-chain emissions related to inputs, all farms - except one were found to be net sources of GHG. The total farm GHG balance varied between a sink of -70 and a source of +310 kg CO_2eq per unit (GJ) energy in animal farm products. Byrne, Kiely and Leahy (2007), measuring C balance for two dairy farms in southwest Ireland, equally considered the farm perimeter as the system boundary for inputs and outputs of C. In the two case studies, both farms appeared as net C sinks, sequestering between 200 and 215 g C/m²/year (Table 9).

Farm-scale mitigation options thus need to be carefully assessed at the production system scale, in order to minimize GHG emissions per unit meat or milk product (Schils *et al.*, 2007). Advanced (Tier 3) and verifiable methodologies still need to be developed in order to include GHG removals obtained by farm-scale mitigation options in agriculture, forestry and land use (AFOLU sector, IPCC 2003) national GHG inventories.



CONCLUSIONS

This review shows that grassland C sequestration is detected both by C stock change (inventories and long-term experiments) and by C flux measurements, however with high variability across studies. Further development of measurement methods and of plot and farm scale models carefully tested at benchmark sites will help further reduce uncertainties. Low-cost mitigation options based on enhancing C sequestration in grasslands are available. Mitigating emissions and adapting livestock production systems to climate change will nevertheless require a major international collaborative effort and the development of extended observational networks combining C (and non CO₂-GHG) flux measurements and long-term experiments to detect C stock changes. C sequestration could play an important role in climate mitigation but, because of its potential reversibility, preserving current soil C stocks and reducing CH₄ and N₂O emissions are strongly needed.

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