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Mitigating the greenhouse gas balance of ruminant production systems through carbon sequestration in grasslands

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Soil carbon 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 $\pm$ 30 g C/m^2 per year according to inventories of SOC stocks and $-$231 and 77 g C/m^2 per 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/m^2 per year for grazed, cut and mixed European grasslands on mineral soils, respectively, 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; (v) converting grass leys to grass-legume mixtures or to permanent grasslands. With nine European sites, direct emissions of N_2O from soil and of CH_4 from enteric fermentation at grazing, expressed in CO_2 equivalents, compensated 10% and 34% of the on-site grassland C sequestration, respectively. Digestion inside the barn of the harvested herbage leads to further emissions of CH_4 and N_2O by the production systems, which were estimated at 130 g CO_2 equivalents/m^2 per year. The net balance of on- and off-site C sequestration, CH_4 and N_2O emissions reached 38 g CO_2 equivalents/m^2 per 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, as soil C sequestration is both reversible and vulnerable to disturbance, biodiversity loss and climate change, CH_4 and N_2O emissions from the livestock sector need to be reduced and current SOC stocks preserved.

Keywords: climate change, CO_2, N_2O, CH_4, soil organic carbon

Implications

The C sequestration potential by grasslands and rangelands could be used to partly mitigate the greenhouse gas (GHG) 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 content. Combined with other mitigation measures, such as a reduction in the use of N fertilisers, of fossil-fuel energy and of N-rich feedstuffs by farms, this may lead to substantial reductions in GHG emissions per unit land area and per unit animal product.

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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 characterised 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 hectares 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% of the value of global agricultural output (Delgado, 2005).

Agriculture accounted for an estimated emission of 5.1 to 6.1 Gigaton (Gt) CO_2 equivalents per year in 2005 (10% to 12% of total global anthropogenic emissions of greenhouse gases (GHGs) (Intergovernmental Panel on Climate Change (IPCC), 2007) and for ca. 60% of N_2O emissions and 50% of CH_4 emissions). Between 1990 and 2005, the direct emissions of the agriculture sector have increased by 17% and this increase has mostly occurred in developing countries (IPCC, 2007). The GHG inventory methodology used by IPCC (IPCC, 1996 and 2006) only includes, however, farm emissions in the agriculture sector. Indirect GHG emissions generated by farm activity through the use of farm inputs (e.g. fertilisers, 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 N fertilisers and of organic pesticides) and transport (e.g. transport of fertilisers and feed). Emissions from electricity and fuel use are covered in the buildings and transport sector, respectively (IPCC, 2006).

Although 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% of global GHG emissions as measured in CO_2 equivalents (Food and Agriculture Organisation (FAO), 2006). Livestock production induces 9% of global anthropogenic CO_2 emissions. The largest share (i.e. 7%) 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% of anthropogenic methane (see Martin *et al.*, 2009) most of that from enteric fermentation by ruminants. Furthermore, it induces 65% of anthropogenic nitrous oxide emissions, the great majority from manure (FAO, 2006).

Agricultural ecosystems hold large C reserves (IPCC, 2001), mostly in soil organic matter. Historically, these systems have lost more than 50 Gt C (Paustian *et al.*, 1998; Lal, 1999 and 2004). Agricultural lands generate very large CO_2 fluxes, both to and from the atmosphere (IPCC, 2001), but the *net* flux would be small (United States-Environmental Protection Agency (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% 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% 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) carbon 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% 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% to 40% 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 maximised by agricultural practices such as frequent grazing and use of highly digestible forage cultivars. Consequently, in these systems the primary factor which 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 *et al.*, 1998); (ii) the role of defoliation intensity and frequency, and of treading by animals, which both reduce the leaf area and then the atmospheric CO_2 capture.

Only a small fraction of the ingested grassland C is accumulated by ruminants in meat production systems (e.g. 0.6% 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% to 20% of C intake; Faverdin *et al.*, 2007). Additional C losses (ca. 3% to 5% of the digestible C) occur through methane emissions from the enteric fermentation (IPCC, 2006; see Martin *et al.*, 2009).

Processes controlling soil organic carbon accumulation
Carbon accumulation in grassland ecosystems occurs mostly below ground. Grassland soils are typically rich in SOC, partly owing to active rhizodeposition (Jones and Donnelly, 2004) and partly to the activity of earthworms that promote macro-aggregate formation in which micro-aggregates form that stabilise SOC for extended periods (Six *et al*., 2002; Bossuyt *et al*., 2005). Rhizodeposition favours C storage (Balesdent and Balabane, 1996), because direct incorporation into the soil matrix allows a high degree of physical stabilisation of the soil organic matter. 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 soil organic matter 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 *et al*., 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.

Sequestred SOC can, if undisturbed, remain in the soil for centuries. In native prairie sites in the US great plains, where SOC was ^{14}C-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 2000 to 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 like 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 after several centuries equilibrium in terms of their C balance (Lal, 2004). Soil C sequestration is reversible, as factors like 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
Carbon sequestration can be determined directly by measuring changes in C pools (Conant *et al*., 2001) and, or, indirectly by measuring C fluxes (Table 1 and equation (1)). SOC stocks display a high spatial variability (coefficient of variation of 50%; Cannell *et al*., 1999) in grassland as compared to 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% ($\pm$4) in dry climates to 29% ($\pm$4) in moist climates. Converting cropland back to grassland uses for 20 years was found to restore 18% ($\pm$7) of the native C stocks in moist climates (relative to the 29% loss owing to long-term cultivation) and 7% ($\pm$5) of native stocks in temperate dry climates (Conant *et al*., 2001). As a result of periodic tillage and resowing, 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, that is, less frequent ploughing (Soussana *et al*., 2004).

Role of management for carbon sequestration in grasslands
A number of studies have analysed effects of grassland and rangeland management on SOC stocks (Table 1). Most studies concern only the top-soil (e.g. 0 to 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 *et al*., 2004). A meta-analysis of 115 studies in pastures and other grazing lands worldwide (Conant *et al*., 2001), indicated that soil C levels increased with improved management (primarily fertilisation, grazing management, and conversion from cultivation or native vegetation, improved grass species) in 74% of the studies considered (Table 1). Light grazing increased SOC stocks compared to exclosure and to heavy grazing (Ganjegunte *et al*., 2005; Table 1). Some of the possible soil C sequestration opportunities for temperate grasslands in France have been calculated and compared (Table 1) for 20-year time periods (Soussana *et al*., 2004). According to these estimates, annual C storage rates between 20 and 50 g C/m^2 per year are obtained for a range of options, which seem compatible with gradual changes in the forage production systems, namely: (i) reducing N fertiliser 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; (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

Table 1 *Literature survey of net C storage (NCS) at grassland sites using different methods: C flux balance (A), grassland soil C inventory (B), soil C change after a change in grassland management (C), and farm scale flux measurements (D). A positive F_{CO_2} represents a net C uptake from the ecosystem. A positive NCS denotes a net carbon accumulation in grassland ecosystems. All fluxes are in g C/m^2 per year*

Grassland type and management	Location	MAT (°C)	MAP (mm)	F_{CO_2} (g C/m^2 per year)	$F_{harvest}$ (g C/m^2 per year)	F_{manure} (g C/m^2 per year)	NCS	Duration (months)	Method	References	Notes
A. Flux balance											
Alpine extensive pasture and hay meadow	Mount Rigi, Central Switerland	8.4	991	−172	183	0	−355	12	Eddy covariance	Rogiers *et al.* (2008)	Drained organic soil
Grazed peat-pasture	Waikato, New Zealand	15	1281	−4.5	619	n.d.	−106	12	Eddy covariance	Nieveen *et al.* (2005)	Drained peat soil
Extensive grazed pasture	East of the Missouri river, North Dakota	15	483	317*a*	n.d.	n.d.	n.d.	10 × 6 months	Bowen Ratio	Phillips and Beeri (2008)	
Extensive grazed pasture	West of the Missouri river, North Dakota	15	390	239*a*	n.d.	n.d.	n.d.	10 × 6 months	Bowen ratio	Phillips and Beeri (2008)	
Extensive grazed pasture	Hungary	10.5	500	69	0	0	68	24	Eddy covariance	Soussana *et al.* (2007)	No N; dry steppe
Extensive grazed pasture	Italy	6.3	1200	360	0	0	358	24	Eddy covariance	Soussana *et al.* (2007)	90 kg N/ha per year
Intensive grassland (grazed and cut)	The Netherlands	10	780	177	220	80	33	12	Eddy covariance	Soussana *et al.* (2007)	300 kg N/ha per year
Intensive grassland (grazed and cut)	Scotland	8.8	638	343	110	3	231	24	Eddy covariance	Soussana *et al.* (2007)	200 kg N/ha per year
Intensive grassland (grazed and cut)	Ireland	9.4	824	293	374	0	−170	24	Eddy covariance	Soussana *et al.* (2007)	200 kg N/ha per year
Intensive meadow (cut)	Denmark	9.2	731	152	333	1400**	1100**	24	Eddy covariance	Soussana *et al.* (2007)	200 kg N/ha per year
Extensive pasture (grazed)	France	7	1200	75	0	0	69	36	Eddy covariance	Allard *et al.* (2007)	No fertilizer
Intensive pasture (grazed)	France	7	1200	99	0	0	87	36	Eddy covariance	Allard *et al.* (2007)	175 kg N/ha per year
Extensive meadow (cut)	Swiss	9.5	1100	254	311	0	−57	36	Eddy covariance	Ammann *et al.* (2007)	No fertilizer
Intensive meadow (cut)	Swiss	9.5	1100	467	368	67.5	147	36	Eddy covariance	Ammann *et al.* (2007)	200 kg N/ha per year
Intensive wetland meadow (grazed and cut)	UK	12.9	750	169	228	0	−34	12	Eddy covariance	Lloyd (2006)	Wet grassland; corrected for animal intake
Intensive grassland (Site A)	County Cork, southern Ireland	10	1470	15	0	n.d.	15**	12	Chamber measurements	Byrne *et al.* (2005)	300 kg N/ha per year. New pasture
Intensive grassland (Site B)	County Cork, southern Ireland	10	1470	38	0	n.d.	38**	12	Chamber measurements	Byrne *et al.* (2005)	300 kg N/ha per year. Permanent pasture
Native tallgrass prairie	North-central Oklahoma, USA	14	1868.5	8	0	0	n.d.	20	Eddy covariance	Suyker and Verma (2001)	Not grazed, prescribed burn
Sparse tussock dry grassland	South Island, New Zealand	9.9	446	−9	0	0	n.d.	24	Eddy covariance	Hunt *et al.* (2004)	Dry year, no N, no burning
Sparse tussock dry grassland	South Island, New Zealand	9.2	933	41	0	0	n.d.	24	Eddy covariance	Hunt *et al.* (2004)	Wet year, no N, no burning
Abandoned moist mixed grassland	Alberta, Canada	15.3	482	109	0	0	n.d.	12	Eddy covariance	Flanagan *et al.* (2002)	1998, wet summer
Abandoned moist mixed grassland	Alberta, Canada	13.2	341	21	0	0	n.d.	12	Eddy covariance	Flanagan *et al.* (2002)	1999, average summer
Abandonned moist mixed grassland	Alberta, Canada	14.5	275.5	−18	0	0	n.d.	12	Eddy covariance	Flanagan *et al.* (2002)	2000, dry summer
Mixed grass	Southeastern Arizona, USA	17	356	−135	0	0	n.d.	48	Bowen ratio	Emmerich (2003)	
Species-rich grassland	UK	n.d.	n.d.	n.d.	n.d.	n.d.	120	48	Chamber measurements	Fitter *et al.* (1997)	Four to five cuts per year

Table 1 *Continued*

Grassland type and management	Location	MAT (°C)	MAP (mm)	F_{CO_2}	$F_{harvest}$ (g C/m^2 per year)	F_{manure}	NCS	Duration (months)	Method	References	Notes
Grazed peat-pasture	California, USA	16.2	1180	28	0	0	n.d.	24	Eddy covariance	Xu and Baldocchi (2004)	
Mixed grass	Mandan ND, USA	n.d.	478	94	0	0	n.d.	4 × 7 months	Bowen ratio	Frank and Dugas (2001)	No fertilizer, no burning, last grazed: 4 years
B. Soil inventories											
Permanent grassland	England, Wales						−5	25 years	Soil C concentration change 0 to 15 cm	Bellamy *et al.* (2005)	
Upland grassland	England, Wales						−37.5	25 years	Soil C concentration change 0 to 15 cm	Bellamy *et al.* (2005)	
Rotational grass	England, Wales						−2.1	25 years	Soil C concentration change 0 to 15 cm	Bellamy *et al.* (2005)	
Grassland	Belgium						44	50 years	Soil C concentration change 0 to 30 cm	Goidts and van Wesemael, (2007)	
Grassland	Belgium						22	40 years	Soil C concentration change 0 to 30 cm	Lettens *et al.* (2005a)	
Grassland	Belgium						−90 (70)	10 years	Soil C concentration change 0 to 100 cm	Lettens *et al.* (2005b)	
Grassland	China						101	18 years	Soil C concentration change	Piao *et al.* (2009)	
C. Management change											
Perennial grassland converted from arable	Central Texas, USA						45	For 6 to 60 years	Soil C stock change 0 to 60 cm	Potter *et al.* (1999)	
Cultivated site to restored grassland	Missouri coteaux, Canada	0.7	320				30 to 290	8 years	Soil C stock change 0 to 30 cm	Nelson *et al.* (2008)	
Heavy to light grazing grassland	Cheyenne, WY, USA	n.d.	384				13.8	21 years	Soil C stock change 0 to 5 cm	Ganjegunte *et al.* (2005)	
Exclosure to light grazing	Cheyenne, WY, USA	n.d.	384				14.3	21 years	Soil C stock change 0 to 5 cm	Ganjegunte *et al.* (2005)	
Nutrients addition via fertilizer	Forty-two data points						30*b*		Soil C stock change	Conant *et al.* (2001)	
Converting cultivated land to grassland	Twenty-three data points						101*b*		Soil C stock change	Conant *et al.* (2001)	
Improved grazing management	Forty-five data points						35*b*		Soil C stock change	Conant *et al.* (2001)	
Improved grass species	Five data points						304*b*		Soil C stock change	Conant *et al.* (2001)	
Restoration of degraded lands	US great plains						80 to 110		Soil C stock change	Follett *et al.* (2001)	
Sown grassland on mineral soil	France						60 to 80		Soil C stock change (organic matter fractions >50 μ)	Loiseau and Soussana (1999)	
Reduction of N fertilizer input	France	9	800				30	10 years	Soil C stock change 0 to 30 cm	Soussana *et al.* (2004)	

Table 1 *Continued*

Grassland type and management	Location	MAT (°C)	MAP (mm)	F_{CO_2} (g C/m^2 per year)	$F_{harvest}$ (g C/m^2 per year)	F_{manure} (g C/m^2 per year)	NCS	Duration (months)	Method	References	Notes
Conversion of short duration grass-ley to grass-legume	France	9	800				30 to 50	10 years	Soil C stock change 0 to 30 cm	Soussana *et al.* (2004)	
Intensification of permanent grassland	France	9	800				20	10 years	Soil C stock change 0 to 30 cm	Soussana *et al.* (2004)	
Intensification of nutrient poor grassland on organic soils	France	7	1100				−100	10 years	Soil C stock change 0 to 30 cm	Soussana *et al.* (2004)	
Permanent grassland to medium duration leys	France	9	800				−20	10 years	Soil C stock change 0 to 30 cm	Soussana *et al.* (2004)	
Increasing the duration of grass leys	France	9	800				20 to 50	10 years	Soil C stock change 0 to 30 cm	Soussana *et al.* (2004)	
Short duration leys to permanent grassland	France	9	800				30 to 40	10 years	Soil C stock change 0 to 30 cm	Soussana *et al.* (2004)	
D. Farm scale											
Intensive grazed and cut grassland	County Cork, southern Ireland	10	1340	290	134	n.d.	205	12	Eddy covariance, farm fluxes	Byrne *et al.* (2007)	300 kg N/ha per year; cattle grazed
Intensive grassland (grazed and cut)	South West Ireland	10	1785	193	70	n.d.	24	12	Eddy covariance, farm fluxes	Jaksic *et al.* (2006)	Wet year, 300 kg N/ha per year
Intensive grassland (grazed and cut)	South West Ireland	10	1185	258	100	n.d.	89	12	Eddy covariance, farm fluxes	Jaksic *et al.* (2006)	Dry year, 300 kg N/ha per year

MAT = mean annual temperature; MAP = mean annual precipitation; F_{CO_2} = net CO_2 ecosystem exchange; F_{manure} = lateral organic C fluxes which are imported (manure application) in the system; $F_{harvest}$ = lateral organic C fluxes which are exported (harvests) from the system; n.d. = not defined.

a average of growing season.

b 87% of the studies were from Australia, the United Kingdom, New Zealand, Canada, Brazil and the United States.

**Not included in mean.

Additional studies can be found in the reviews by Conant *et al.* (2001) and by Ogle *et al.* (2004).

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/m^2 per 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 ca. 10 g C/m^2 per year (Table 1). The Countryside Surveys of Great Britain are ongoing ecological assessments in UK that have taken place since 1978 (Firbank *et al.*, 2003). In this survey, significant increases in soil C concentration, ranging from 0.2 to 2.1 g/kg per 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/m^2 per year (Lettens *et al.*, 2005a; Goidts and van Wesemael, 2007, respectively), or losing C at 90 g C/m^2 per 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, as 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 Megatons SOC sequestration per year on 3.5 billion ha of permanent pasture worldwide. Using national grassland resource dataset and NDVI (Normalised Difference Vegetation Index) 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/m^2 per 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 to forests and croplands. They estimated a net grassland C sink of 66 $\pm$ 90 g C/m^2 per year over geographic Europe, though 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 owing to changes in stocks in above and below-ground biomass; (ii) emissions of non-CO_2 GHGs (CO, CH_4, N_2O and NO_x) owing to biomass burning (Fearnside, 2000); and (iii) C emissions and removals in grasslands owing to changes in soil C stocks. Mineral and organic soils (peat, histosoils, etc.) are separated for the calculations of soil C stock changes, provided that national inventory data are available for grassland soils (IPCC, 2006). 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% 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% in temperate and tropical regions, respectively, and with an additional improvement(s), stocks increased by another 11%. By applying these factors to managed grasslands in the USA, Ogle *et al.* (2004) found that over a 20-year period, changing management could sequester up to 142 Megatons C per 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 C 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. CO_2; CH_4; volatile organic compounds, VOC; and emissions during fires), (ii) organic C imports (manures) and exports (harvests, animal products), (iii) dissolved C lost in waters (dissolved organic and inorganic C) and lateral transport of soil C through erosion (Figure 1). NCS (g C/m^2 per year) is the

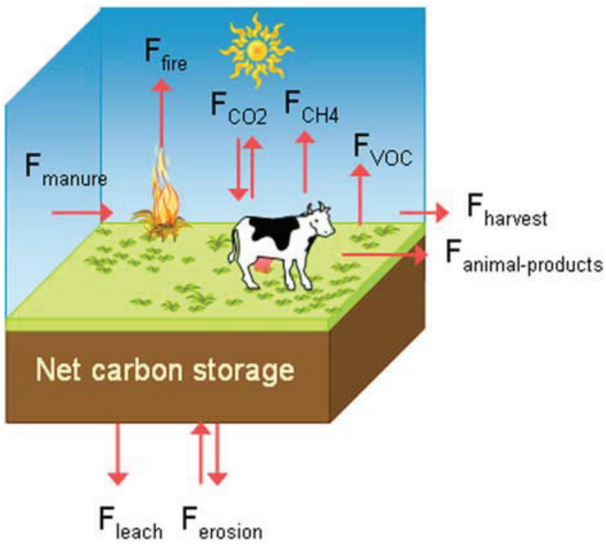

Figure 1 Carbon fluxes (g C/m^2 per year) in a managed grassland. F_{CO_2} is the net CO_2 ecosystem exchange. F_{fire} is the total C loss by fire; F_{CH_4}, F_{VOC} are non-CO_2 trace gas C losses from the ecosystem, as methane and volatile organic carbon, respectively. F_{manure}, $F_{harvest}$ and $F_{animal\text{-}products}$ are lateral organic C fluxes, which are either imported (manure application) or exported (harvests and animal products) from the system. F_{leach} and $F_{erosion}$ are organic (and/or inorganic) C losses through leaching and erosion, respectively. Net carbon storage (NCS, see equation (1)) is calculated as the balance of carbon fluxes.

mass balance of these fluxes (equation (1)):

$$NCS = (F_{CO_2} - F_{CH_4-C} - F_{VOC} - F_{fire}) + (F_{manure} - F_{harvest} - F_{animal\text{-}products}) - (F_{leach} + F_{erosion}), \quad (1)$$

where F_{CO_2} is equal to the net ecosystem exchange (NEE) of CO_2 between the ecosystem and the atmosphere, which is conventionally positive for a C gain by the ecosystem. F_{CH_4-C}, F_{VOC} and F_{fire} are trace gas C losses from the ecosystem (g C/m^2 per year). F_{manure}, $F_{harvest}$ and $F_{animal\text{-}products}$ are lateral organic C fluxes (g C/m^2 per 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/m^2 per 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 like Europe (i.e. below 1 g C/m^2 per year over 1997 to 2004), while they reach 10 and 100 g C/m^2 per year in the 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/m^2 per 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/m^2 per 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):

$$NCS = (F_{CO_2} - F_{CH_4-C}) + (F_{manure} - F_{harvest} - F_{animal\text{-}products}) - F_{leach}. \quad (2)$$

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. As 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' carbon, 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/m^2 per year) to significant release (160 g C/m^2 per 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 ($F_{CO_2} < 0$) 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 (Nieveen *et al*., 2005; Rogiers *et al*., 2008), and these sites were found to lose C (i.e. negative NCS; Table 1).

Within the European (FP5 EESD) 'GreenGrass' project, the full GHG balance of nine contrasted grassland sites covering a major climatic gradient over Europe (Tables 1 and 2), 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 nitrogen fertiliser supplies. Two sites (in Ireland and in Switzerland; Table 1) were sown grass-legume mixtures, while the remainder were long-term grasslands. At all sites, the NEE of CO_2 was assessed using the eddy covariance technique. CH_4 emissions resulting from enteric fermentation of the grazing cattle were measured *in situ* at four sites using the SF_6 tracer method. N_2O 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/m^2 per 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 2. Results, corrected for animal exports ($F_{animal\text{-}products}$) and for C leaching ($F_{leach}$), show that NCS varied between 50 and 129 g C/m^2 per year and was higher in grazed than in cut grasslands (Figure 2). Across sites, NCS declined with the degree of herbage utilisation 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 (Allard *et al*., 2007; Klumpp *et al*., 2007). The uncertainty associated to NCS can be estimated using Gaussian error propagation rules and accounting for site number in each management type. NCS uncertainty reached 25% and 80% 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/m^2 per year according to inventories of SOC stocks and 22 ± 56 g C/m^2 per year according to C flux balance (Table 1). 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/m^2 per year, respectively, Table 1A) and inventory (Bellamy *et al*., 2005)

Table 2 *Mean annual greenhouse fluxes in CO_2 equivalents/m^2 per year of managed European grassland sites studied by Soussana* et al. *(2007)*

Management	NCS	Att-NCS	Grassland methane $GWP_{CH_4}F_{CH_4}$	Total methane $GWP_{CH_4}F_{CH_4}$	Grassland N_2O $GWP_{N_2O}F_{N_2O}$	Total N_2O $GWP_{N_2O}F_{N_2O}$	NGHG	Att-NGHG
Grazing	471	471	145	145	22	22	320	320
Grazing and cutting	183	268	159	476	64	81	−22	−272
Cutting	259	359	0	447	30	53	230	−141

NCS = net carbon storage in the grassland (see equation (2)); Att-NCS = attributed net carbon storage (see equation (4)); NGHG = net greenhouse gas balance (see equation (3)); Att-NGHG = attributed net greenhouse gas balance (see equation (5)); GWP = global warming potential.
Data are means of two, four and three European sites for grazed only (meat production systems), cut and grazed (meat and dairy production systems), and cut only (dairy production systems) grasslands.
A positive value of NCS, Att-NCS, NGHG and Att-NGHG denotes a sink activity of the grassland ecosystems.

methods, organic soils would be more susceptible of 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 *v.* permanent, Byrne *et al.*, 2005), N fertiliser supply (Ammann *et al.*, 2007), grazing pressure (Allard *et al.*, 2007), drainage (Nieveen *et al.*, 2005; Rogiers *et al.*, 2008) and burning (Suyker and Verma, 2001) (Table 1). 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 fertilisation, reduced herbage utilisation by grazing and cutting), which have carry-over effects on soil C pools. In addition, the recent rise in air temperature, in atmospheric CO_2 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 GHGs (Robertson *et al.*, 2000). N_2O and CH_4 emissions are often expressed in terms of CO_2 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. For example, over the 100-year timescale, on a kilogram-for-kilogram basis, one unit of nitrous oxide has the same global warming potential (GWP) as 298 units of carbon dioxide ($GWP_{N_2O} = 298$), whereas one unit of methane has the same GWP as 25 units of carbon dioxide ($GWP_{CH_4} = 25$) (IPCC, 2007). An integrated approach is needed to quantify in CO_2 equivalents the fluxes of all the three trace gases (CO_2, CH_4 and N_2O).

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 N_2O and CH_4 emissions at farm scale. As agricultural management is one of the key drivers of the sequestration and emission processes, for grasslands there is potential to reduce the net GHG flux, expressed in CO_2 equivalents. Methane emissions by enteric fermentation under grazing conditions are reviewed in details by Martin *et al.* (2009). Below, we focus on N_2O emissions and on the GHG balance in CO_2 equivalents.

Nitrous oxide emissions from grassland soils

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. 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.

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 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 (Skiba and Smith, 2000).

For given soil and climate conditions, N_2O emissions are likely to scale with the nitrogen fertiliser inputs. Therefore, the current IPCC (2006) methodology assumes a default emission factor (EF_1) of 1% (ranging from 0.3% to 3%) for non-tropical soils emitted as N_2O per unit nitrogen input N (0.003 to 0.03 kg N_2O-N/kg N input).

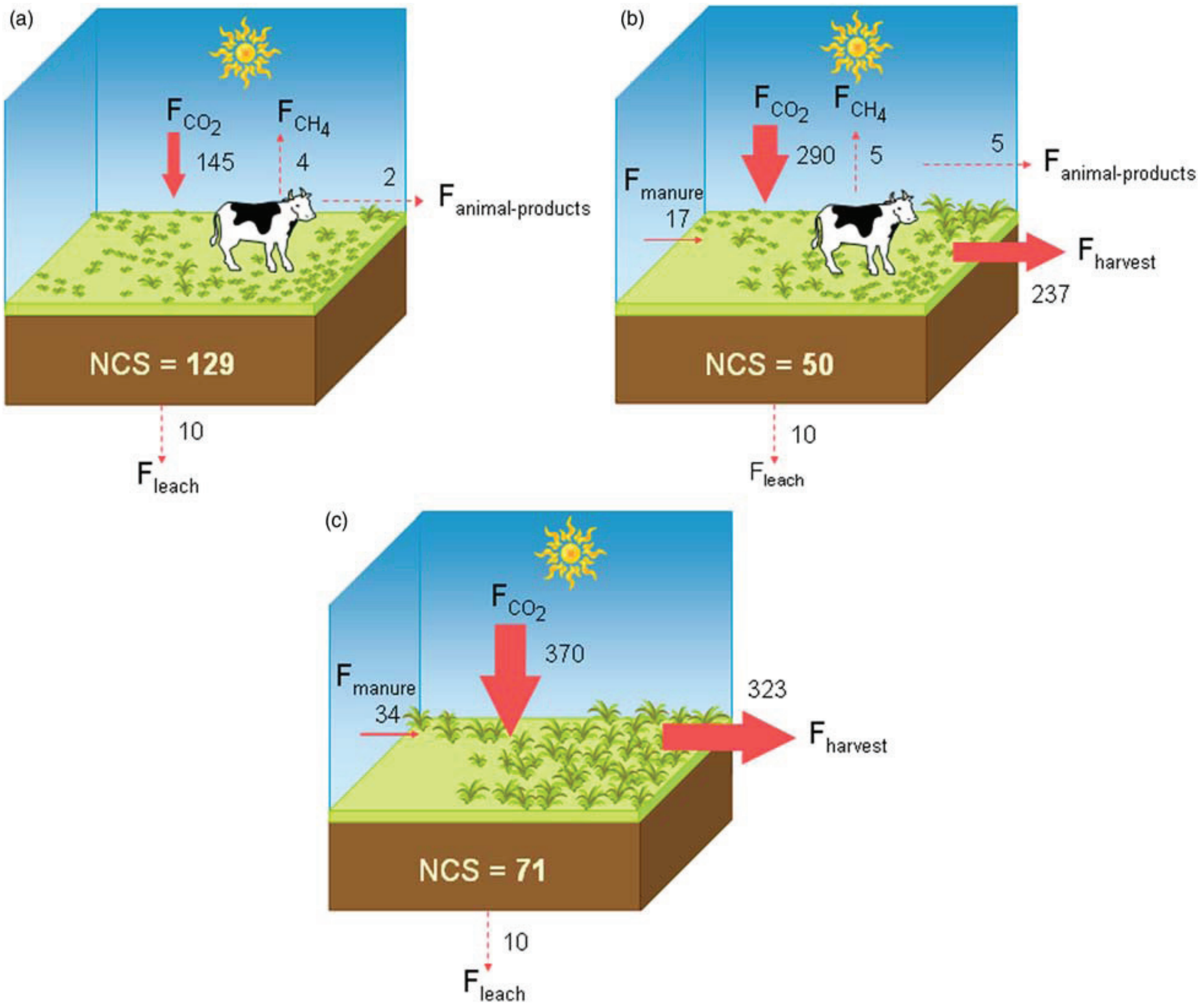

Figure 2 Carbon fluxes (g C/m^2 per year) in managed European grassland systems studied by Soussana *et al.* (2007). Net carbon storage in the grassland (NCS, see equation (2)) in (a) grazed only, (b) cut and grazed and (c) cut only grasslands is calculated as the balance of carbon fluxes. For abbreviations, see Figure 1. Data are means of two, four and three European sites for grazed only ((a), meat production systems), cut and grazed ((b), meat and dairy production systems) and cut only ((a), dairy production systems) grasslands. A standard F_{leach} value (10 g C/m^2 per year) was assumed for all sites. C exports in animal products were assumed to reach 2% and 20% of C intake for meat and milk production, respectively (see text). Grazed sites: Hungary, France, Italy (see Allard *et al.*, 2007; Soussana *et al.*, 2007; Table 1). Cut and grazed sites: Scotland, Ireland and the Netherlands (see Soussana *et al.*, 2007; Table 1). Cut sites: Switzerland (see Ammann *et al.*, 2007; Table 1). A positive value of NCS and attributed NCS denotes a sink activity of the grassland ecosystem.

N_2O emissions in soils usually occur in 'hot spots' associated with urine spots and particles of residues and fertiliser, despite the diffused spreading of fertilisers and manure (Flechard *et al.*, 2007). Nitrous oxide emissions from grasslands also tend to occur in short-lived bursts following the application of fertilisers (Clayton *et al.*, 1997; Leahy *et al.*, 2004). 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% 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 fertilised systems were highly variable, but the mean emission factor (0.75%) was substantially lower than the IPCC default value of 1.0% for direct emissions of N_2O from N fertilisers (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% to 5% (Crutzen *et al.*, 2007). This factor is covered only in part by the 1% of direct emissions factor. Additional indirect emissions, resulting from further N_2O emissions at the landscape scale, are also accounted for by IPCC (2006).

Methane exchanged with grassland 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 at a larger rate than cropland soil (6 and 3 kg CH_4/ha per 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 methane emissions. In an abandoned peat meadow, methane emissions were lower in water-unsaturated compared to water-saturated soil conditions (Hendriks *et al*., 2007). Keppler *et al*. (2006) have shown the emissions of low amounts of CH_4 by terrestrial plants under aerobic conditions. However, this claim has not been confirmed since 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_{CH_4-C} and F_{N_2O}) of non-CO_2 radiatively active trace gases and calculate a net exchange rate in CO_2 equivalents (net greenhouse gas balance, NGHG; g CO_2/m^2 per year, equation (3)), using the GWP of each gas at the 100-years time horizon (IPCC, 2007):

$$NGHG = k_{CO_2}(NCS + F_{CH_4-C}) - GWP_{CH_4}F_{CH_4} - GWP_{N_2O}F_{N_2O}, \quad (3)$$

where $k_{CO_2} = 44/12$ g CO_2-g C, F_{CH_4} is the methane emission (g CH_4/m^2 per year) and F_{N_2O} is the nitrous oxide emission (g N_2O/m^2 per year). CH_4 is not double counted as CO_2 in equation (4), as F_{CH_4-C} is added to NCS.

On average, of the nine sites covered by the 'GreenGrass' European project, the grassland plots displayed annual N_2O and CH_4 emissions of 39 and 101 g CO_2 equivalents/m^2 per year, respectively (Table 2). Hence, when expressed in CO_2 equivalents, emissions of N_2O and CH_4 compensated 10% and 34% of the on-site grassland C sequestration, respectively. The mean on-site NGHG reached 198 g CO_2 equivalents/m^2 per year, indicating a sink for the atmosphere. Nevertheless, sites that were intensively managed by grazing and cutting had a negative NGHG and were therefore estimated to be GHG sources in CO_2 equivalents.

Vulnerability of soil organic carbon to climate change

Although the ancient carbon located in the deep soil is presumably protected from microbial decomposition by a lack of easily degradable substrates (Fontaine *et al*., 2003), soil C stocks in grassland ecosystems are vulnerable to climate change. The 2003 heat wave and drought reduced by 30% the total gross primary productivity over Europe, which resulted in a strong anomalous net source of carbon dioxide (0.5 Gt C per year) to the atmosphere and reversed the effect of 4 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 carbon-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 CO_2 reduces the sensitivity of grassland ecosystems to drought (Morgan *et al*., 2004) and increases grassland productivity by 5% to 15% depending on water and nutrients availability (Soussana and Hartwig, 1996; Soussana *et al*., 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 carbon model and using climate data from four global climate models implementing four IPCC emission scenarios. 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 to 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 CO_2, 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 *et al*., 2005).

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

The role of grassland carbon 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 1D), but often neglected, role in the farm GHG budget.

Carbon transfer between different fields is very common in livestock production systems. The application of organic manure to certain fields may also strongly vary 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 carbon in the harvested herbage returns within one year to the atmosphere. This is usually not the case, as the non-digestible carbon 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 that 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 methane 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 carbon 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 to the amount of manure it receives (Figure 3). On-site decomposition of manure C supplied to the studied grassland field contributes to ecosystem respiration (F_{CO_2}; equation (2)) and is therefore not double counted as an off-site CO_2 flux.

Off-site C sequestration ($NCS_{@barn}$) 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:

$$\text{Att-NCS} = \text{NCS} + \text{NCS}_{@barn} = \text{NCS} + f_{humif}.\text{Max}[0, (1 - f_{digest})F_{harvest} - F_{manure}] - F_{CH_4 manure\_C}, \quad (4)$$

where f_{humif} is the fraction of non-labile C in manure, f_{digest} is the proportion of ingested C that is digestible and $F_{CH_4 manure\_C}$ is methane emission from farm effluents calculated according to IPCC (2006) Tier 2 method in CO_2-C equivalents (Figure 3). The fraction of non-labile C in manure (f_{humif}) varies between 0.25 and 0.45 (Soussana *et al.*, 2004).

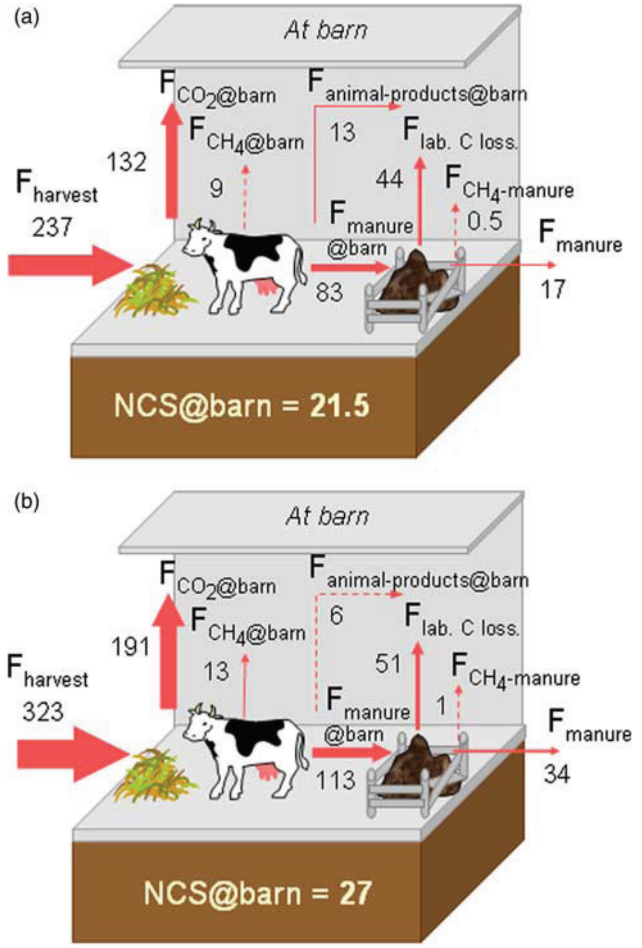

Figure 3 Carbon fluxes (g C/m^2 per year) in managed European grassland systems studied by Soussana *et al.* (2007). Net carbon storage in the barn ($NCS_{@barn}$) in (a) cut and grazed and (b) cut only grasslands are calculated as the balance of carbon fluxes. $F_{CO_2@barn}$, $F_{animal\text{-}products@barn}$ and $F_{labile\text{-}C\text{-}losses}$ are, respectively, CO_2 emissions, C exports in animal products from ruminants and CO_2 losses from microbial degradation of farm effluents during storage and after spreading. $F_{CH_4@barn}$ and $F_{CH_4\text{-}manure}$ are the CH_4 emissions at barn from enteric fermentation and farm effluents, respectively. For other abbreviations, see Figure 1. Carbon fluxes at barn were estimated assuming the same type of production (meat or milk) in the barn and in the grassland and solid manure (see equation (4)). C exports in animal products at barn were assumed to be 2% and 20% of C intake for meat and milk production, respectively (see text).

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, as 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.

$NCS_{@barn}$ reached 21.5 and 27 g C/m^2 per year for mixed and cut systems, respectively. Therefore, Att-NCS, which includes $NCS_{@barn}$, was higher in grazed (129 g C/m^2 per year) than in cut and mixed grassland systems (98.5 and 71 g C/m^2 per year, respectively). These estimates do not include C emissions from machinery, which are higher in cut (e.g. mowing, silage making) compared to grazed systems, but are not part of the AFOLU (Agriculture, Forestry and Other Land Uses) sector and are not discussed in this review.

GHG balance during housing

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. 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).

The fraction of non-labile C (f_{humif}) 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% of C initially present in composted and anaerobically stored manure, respectively (Thomsen and Olesen, 2000).

In order to account for: (i) the offsite CO_2, CH_4 and N_2O emissions resulting directly from the digestion by cattle of the forage harvests, (ii) the contribution to CH_4 and N_2O emissions by farm effluents and (iii) the manure and slurry applications which add organic C to the soil, an attributed net GHG balance (Att-NGHG) was adapted from Soussana *et al.* (2007) as:

$$\begin{aligned}\text{Att-NGHG} = k_{CO_2}(\text{Att-NCS} + F_{CH_4\text{-}C}) - GWP_{CH_4}(F_{CH_4} \\ + F_{CH_4@barn} + F_{CH_4\text{-manure}}) - GWP_{N_2O}(F_{N_2O} \\ + F_{N_2O\text{-manure}}),\end{aligned} \tag{5}$$

where $F_{CH_4@barn}$ is CH_4 emission by enteric fermentation at barn (g CH_4/m^2 per year), $F_{CH_4\text{-manure}}$ (g CH_4/m^2 per year) and $F_{N_2O\text{-manure}}$ (g N_2O/m^2 per year) are the CH_4 and direct N_2O emissions from farm effluents, respectively, which were calculated according to IPCC (2006) Tier 2 method (Table 2).

Estimated methane emissions at barn from cut herbage reached up to 447 g CO_2 equivalents/m^2 per year (Table 2) 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 2). Therefore, a grazing management seems to be a better strategy for removing GHG from the atmosphere than a 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 utilisation of the cut herbage by ruminants 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; (ii) the lack of detailed grassland soil C inventories for soil model initialisation (Vuichard *et al.*, 2007a).

Including carbon sequestration in greenhouse gas budgets at farm scale

A grazing livestock farm consists in a productive unit that converts various resources into outputs as milk, meat and sometimes grains too. In Europe, many ruminant farms have mixed farming systems: they produce themselves the roughage and, most often, part of the animal's feeds 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 fertilisers and feed, and they always use direct energy derived from fossil fuels. The net emissions of GHGs (methane, nitrous oxide and carbon dioxide) are related to C and nitrogen flows and to environmental conditions.

To date, only few recent models has been developed to estimate the farm GHG balance (Schils *et al.*, 2007). Most models have used fixed emission factors both for indoors and outdoors 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 net GHG balance at the farm gate is calculated in CO_2 equivalents. Emissions induced by the production and transport of farm inputs (fuel, electricity, N fertilisers 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 – but 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_2 equivalents per unit (GJ) energy in animal farm products. Byrne *et al*. (2007), measuring C balance for two dairy farms in South West 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/m^2 per year (Table 1).

Farm scale mitigation options thus need to be carefully assessed at the production system scale, in order to minimise 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, 2006) 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-CO_2-GHG) flux measurements and long-term experiments to detect C stock changes. Carbon sequestration could play an important role in climate mitigation, but because of its potential reversibility preserving current soil C stocks and reducing CH_4 and N_2O emissions is strongly needed.

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