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Full accounting of the greenhouse gas (CO_2, N_2O, CH_4) budget of nine European grassland sites

J.F. Soussana [a,1,*], V. Allard [a,1], K. Pilegaard [b], P. Ambus [b], C. Amman [c], C. Campbell [d], E. Ceschia [a,2], J. Clifton-Brown [e,3], S. Czobel [f], R. Domingues [g], C. Flechard [c], J. Fuhrer [c], A. Hensen [h], L. Horvath [j], M. Jones [e], G. Kasper [g], C. Martin [i], Z. Nagy [f], A. Neftel [c], A. Raschi [k], S. Baronti [k], R.M. Rees [l], U. Skiba [d], P. Stefani [m], G. Manca [j], M. Sutton [d], Z. Tuba [f], R. Valentini [m]

[a] *INRA, UR874 Grassland Ecosystem Research, F-63100 Clermont-Ferrand, France*
[b] *Risoe National Laboratory, Denmark*
[c] *Agroscope FAL, Air Pollution/Climate Group, Zurich, Switzerland*
[d] *CEH, Edinburgh Research Station, UK*
[e] *Trinity College, Dublin, Ireland*
[f] *Szent István University, Gödöllö, Hungary*
[g] *Wageningen University and Research, Wageningen, The Netherlands*
[h] *ECN, Petten, The Netherlands*
[i] *INRA, UR1213 Herbivores, F-63122 St Genès Champanelle, France*
[j] *Hungarian Meteorological Service, Hungary*
[k] *CNR-IATA, Firenze, Italy*
[l] *Scottish Agricultural College, Edinburgh, UK*
[m] *University of Tuscia, Italy*

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Abstract

The full greenhouse gas balance of nine contrasted grassland sites covering a major climatic gradient over Europe was measured during two complete years. 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 fertilizer supplies. At all sites, the net ecosystem exchange (NEE) of CO_2 was assessed using the eddy covariance technique. N_2O emissions were monitored using various techniques (GC-cuvette systems, automated chambers and tunable diode laser) and CH_4 emissions resulting from enteric fermentation of the grazing cattle were measured *in situ* at four sites using the SF_6 tracer method. Averaged over the two measurement years, net ecosystem exchange (NEE) results show that the nine grassland plots displayed a net sink for atmospheric CO_2 of -240 ± 70 g C m^{-2} year^{-1} (mean $\pm$ confidence interval at $p > 0.95$). Because of organic C exports (from cut and removed herbage) being usually greater than C imports (from manure spreading), the average C storage (net biome productivity, NBP) in the grassland plots was estimated at -104 ± 73 g C m^{-2} year^{-1}, that is 43% of the atmospheric CO_2 sink. On average of the 2 years, the grassland plots displayed annual N_2O and CH_4 (from enteric fermentation by grazing cattle) emissions, in CO_2-C equivalents, of 14 ± 4.7 and 32 ± 6.8 g CO_2-C equiv. m^{-2} year^{-1}, respectively. Hence, when expressed in CO_2-C equivalents, emissions of N_2O and CH_4 resulted in a 19% offset of the NEE sink activity. An attributed GHG balance has been calculated by subtracting from the NBP: (i) N_2O and CH_4 emissions occurring within the grassland plot and (ii) off-site emissions of CO_2 and CH_4 as a result of the digestion and enteric fermentation by cattle of the cut herbage. On average of the nine sites, the attributed GHG balance was not significantly different from zero (-85 ± 77 g CO_2-C equiv. m^{-2} year^{-1}).

* Corresponding author. Tel.: +33 4 73 62 44 23; fax: +33 4 73 62 44 57.
E-mail address: soussana@clermont.inra.fr (J.F. Soussana).
[1] The first two authors have contributed equally to this work.
[2] Present address: CESBIO, Toulouse, France.
[3] Present address: IGER, Aberystwyth, UK.

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The net exchanges by the grassland ecosystems of CO_2 and of GHG were highly correlated with the difference in carbon used by grazing versus cutting, indicating that cut grasslands have a greater on-site sink activity than grazed grasslands. However, the net biome productivity was significantly correlated to the total C used by grazing and cutting, indicating that, on average, net carbon storage declines with herbage utilisation for herbivores.

Keywords: Carbon sequestration; Nitrogen cycle; Nitrous oxide; Methane; Livestock

1. Introduction

Grassland is one of the dominant land uses in Europe, covering 80 million ha that is 22% of the EU-25 land area (EEA, 2005). Most grassland in Europe are managed for feeding domestic herbivores, either directly at grazing or through forage production which is stored as hay or silage. Grasslands contribute to the biosphere–atmosphere exchange of radiatively active trace gases, with fluxes intimately linked to management practices. Of the three greenhouse gases that are exchanged by grasslands, CO_2 is exchanged with the soil and vegetation, N_2O is emitted by soils and CH_4 is emitted by livestock at grazing and can be exchanged with the soil (Soussana et al., 2004).

For grasslands, the nature, frequency and intensity of disturbance plays a key role in the C balance. In a cutting regime, a large part of the primary production is exported from the plot as hay or silage, but part of these C exports may be compensated for by farm manure and slurry application. The largest part of the organic carbon ingested during grazing is digestible (up to 75% for highly digestible forages) and, hence, is respired shortly after intake. Only a small fraction is accumulated in the body of domestic herbivores or is exported as milk. Large domestic herbivores, such as cows, respire approximately 1 tonne of carbon per year (Vermorel, 1995).

Additional carbon losses (ca. 5% of the digestible carbon) occur through methane emissions from enteric fermentation. The non-digestible carbon (from 25 to 40% of the intake depending on the digestibility of the grazed herbage) is returned to the pasture in excreta (mainly as faeces). In most European husbandry systems, the herbage digestibility tends to be maximised by agricultural practices such as frequent grazing and use of highly digestible forage cultivars. Consequently, the primary factor which modifies the carbon flux returned to the soil by excreta is the grazing pressure which varies with the annual stocking rate (mean number of livestock units per unit area). Secondary effects of grazing on the carbon cycle of a pasture include: (i) the role of excretal returns, concentrated in patches, for the SOM mineralisation and the N cycling, especially in nutrient-poor grasslands and (ii) the role of defoliation by animals and of treading, both of which reduce the leaf area and canopy photosynthesis.

Managed European grasslands are often fertilized to sustain productivity and thus emit N_2O to the atmosphere above the background level that is found in natural systems (Jarvis et al., 2001). Typical N_2O emissions from grassland soils, converted into CO_2 equivalent on a 100-year time horizon (Bouwman, 1996) range between 100 and 1000 kg CO_2-C equiv. ha^{-1} $year^{-1}$ (Machefert et al., 2002; Sozanska et al., 2002). One recent estimate of N_2O fluxes from grasslands indicates a mean emission of 2.0 kg N_2O-N ha^{-1} $year^{-1}$, which translates into 250 kg CO_2-C equiv. ha^{-1} $year^{-1}$ (Freibauer et al., 2004).

There are only few continental scale modelling estimates of the GHG budget of grasslands, primarily focused on the CO_2 component of the GHG budget. Vleeshouwers and Verhagen (2002), further quoted by Janssens et al. (2003), applied a semi-empirical model of land use induced soil carbon disturbances to the European continent (as far east as the Urals) and inferred a carbon sink of 101 tonnes g C $year^{-1}$ over grasslands (520 kg C ha^{-1} $year^{-1}$) with uncertainties above the mean.

Currently, the net global warming potential (in terms of CO_2 equivalent) from the greenhouse gas exchanges with European grasslands is not known, because there have been very few direct and long-term measurements of the fluxes.

Table 1
Location and main climate characteristics of the sites in the GREENGRASS network

Acronym	Site name	Country	Latitude	Longitude	Elevation (m a.s.l.)	Mean annual rainfall (mm)	Mean air temperature (°C)	MTD[a] (°C)	Days per year above 5 °C
BG	Bugac	Hungary	46°41′N	19°36′E	140	500	10.5	27.0	235
BS	Easter Bush	Scotland	55°52′N	3°2′W	190	638	8.8	12.1	305
CA	Carlow	Ireland	52°52′N	6°54′W	56	824	9.4	11.3	305
LA	Laqueuille	France	45°38′N	2°44′E	1040	1313	8	18.3	243
LE	Lelystad	The Netherlands	52°30′N	5°30′E	−5	780	10	18.8	266
MA	Malga Arpaco	Italy	46°07′N	11°42′E	1699	1200	6.3	22.0	178
OE	Oensingen	Switzerland	47°17′N	07°44′E	450	1109	9	25.4	244
LV	Lille Valby	Denmark	55°41′N	12°07′E	15	731	9.2	21.3	249

[a] MTD, maximum temperature difference between average monthly means.

An integrated approach, that would allow the simultaneous quantification of all three radiatively active trace gases (CO_2, CH_4 and N_2O), would be desirable as management choices to reduce emissions involve potential trade-offs. For example, improving the primary productivity of grasslands by N fertilizer supply may favour below-ground C storage but is also likely to lead to increased N_2O and CH_4 emissions (Vuichard et al., 2007).

A network of grassland sites was recently established as part of the GREENGRASS [European Commission DG Research 5th Framework Programme—Contract no. EVK2-CT2001-00105] project. Nine grassland sites along a major Europe wide transect have been equipped to measure the net exchange of greenhouse gases (CO_2, N_2O and CH_4) with the atmosphere, using eddy covariance for CO_2, static chambers and eddy correlation for N_2O and the *in situ* SF_6 tracer technique (Johnson et al., 1994; Pinares-Patiño et al., 2007) for the emission of CH_4 by herbivores at grazing.

We present here the results obtained from 2 years of measurements in this site network and assess in CO_2-C equivalents the net radiative forcing resulting from the exchanges with the atmosphere of CO_2, N_2O and CH_4.

2. Materials and methods

2.1. Study sites

The site network covers a NW to SE gradient in Europe with sites ranging from Scotland and Denmark to Italy and Hungary. Along this gradient, the mean annual temperature falls within a narrow range (between 8 and 10.5 °C) for most sites, with the exception of the mountain site of Malga Alparco (1699 m in Italy, 6.3 °C) (Table 1). Atlantic sites display a low contrast between summer and winter temperatures compared to more continental sites. The difference between the minimal and maximal monthly means spans from 11 °C in Carlow (Ireland) to 27 °C in Bugac Puszta (Hungary). The average number of days in the year with a daily mean above 5 °C ranges between 178 days in the mountain climate of Malga Alparco and 305 days in Scotland and Ireland (Easter Bush and Carlow). The mean annual precipitation also contrasts between dry continental sites (500 mm in Hungary), temperate Atlantic sites (between 700 and 900 mm) and wet mountain sites (above 1000 mm in the Alps and in the French Massif Central) (Table 1).

The network includes three main grassland types (Table 2): (i) semi-natural permanent grasslands which are only grazed by cattle without being cut (Bugacpuszta BG, Laqueuille LA and Malga Alparco, MA), (ii) intensively managed permanent grasslands which are used both by grazing and by cutting and (iii) recently sown grass-clover swards (Oensingen OE and Carlow CA) which are cut only (OE) or used by grazing and cutting (CA). In addition, a grass-crop rotation was investigated at the Lille Valby (LV) site, which was sown with a barley crop in 2002 and with a grass undercover of barley in 2003.

Total N fertilizer supply varies from 0 up to 300 kg N ha^{-1} $year^{-1}$ in the site network (Table 2), reflecting typical agricultural management for each site. N was supplied both through mineral and through organic fertilizer applications, the latter adding some organic carbon to the grassland plots. At two of the sites (Laqueuille and Oensingen), two management treatments (intensive, with and extensive without N supply) were compared by replicating all measurements in each treatment.

2.2. CO_2 fluxes

Each site was equipped with an eddy covariance system in spring 2002 for the measurement of the net ecosystem exchange (NEE) of CO_2. All sites, except MA, started recording CO_2 fluxes between May and July 2002. The measurements in MA were delayed until December 2002. The eddy covariance system consisted of a fast response 3D sonic anemometer coupled with fast CO_2–H_2O analyzers (open path in BG, BS, LV, OE, LA and MA and closed path in CA and LE) measuring fluxes of CO_2, latent heat, sensible heat and momentum at a 30 min time step. Details of the materials and software used in each site are provided by Gilmanov et al. (2007). Flux calculations, corrections

Table 2
Grassland types and management at sites in the GREENGRASS network

Site[a]	Grassland/crop type	Management	Total N fertilization (kg N ha^{-1} $year^{-1}$)	Type of organic N fertilization
BG	Semi-natural grassland	Grazing	0	–
BS	Intensive permanent grassland	Grazing and cutting	200	–
CA	Sown grass/clover	Grazing and cutting	200	–
LA intensive	Semi-natural grassland	Grazing	175	–
LA extensive	Semi-natural grassland	Grazing	0	–
LE	Intensive permanent grassland	Grazing and cutting	300	Cattle slurry
MA	Semi-natural grassland	Grazing	90	–
OE intensive	Sown grass/clover	Cutting	200	Cattle slurry
OE extensive	Sown grass/clover	Cutting	0	–
LV	Barley–grass rotation	Cutting	200	Horse manure

[a] For abbreviations see Table 1.

and quality checks of the data were done following Aubinet et al. (2000) by each site manager. An additional filtering criterion was applied based on friction velocity (u^*) values. Critical thresholds of u^* below which CO_2 flux was strongly dependent of u^* was determined for each site (see Gilmanov et al., 2007). Data gaps caused by inadequate quality or sensor failure were reconstructed for all sites using the same gap-filling strategy (Reichstein et al., 2005). In order to calculate an annual CO_2 budget for two successive years, gap-filled data over two successive 1-year periods were used. Given the differences in the CO_2 measurement starting dates, these time periods varied between sites (see Table 4). In the LE site, as a result of a failure in the equipment, no flux data were available in the first year during 5 months and the annual NEE could not be calculated.

2.3. N_2O emissions

N_2O emissions from soils were measured at all sites over the experimental period with both static chambers and with tunable diode laser equipments. More details about materials and methods used for N_2O measurements in the GREENGRASS network are provided by Flechard et al. (2007). The annual N_2O budget (E_{N_2O}, mg N_2O m^{-2} year^{-1}) for each site was calculated according to Flechard et al. (2007).

2.4. CH_4 emissions from enteric fermentation

CH_4 emissions from the enteric fermentation by grazing cattle were measured in five of the grazed sites (CA, BS, LAi, LAe and LE) using the SF_6 tracer technique (Johnson et al., 1994). In order to reduce the variability in methods between sites, all the materials required to make these measurements were build in the same laboratory (INRA, Theix). In the Easter Bush (BS) site, the measurements could not be performed directly on the privately managed site and were therefore carried out on a similar pasture within the same area.

The annual CH_4 budget (E_{CH_4} in g CH_4 ha^{-1} year^{-1}) of CA, LAi, LAe and LE was calculated for each of the two time periods as

$$E_{CH_4} = k_{CH_4}\,\mathrm{SR}\,W \tag{1}$$

where k_{CH_4} is the CH_4 emission rate in g CH_4 kg^{-1} LW day^{-1}; SR the mean annual animal stocking rate in heads per hectare; W is the mean annual liveweight per head in kilograms.

The CH_4 budget at the three remaining grazed sites (BS, BG and MA), was not measured directly on-site but was estimated from measured values of k_{CH_4}. The k_{CH_4} value determined off-site for non-lactating cows at BS was applied to this site, neglecting the small contribution of sheep (less than 5% of the total heads) to the methane emissions of this site. The k_{CH_4} value determined in the extensively managed semi-natural grassland of the Laqueuille site (LAe) was applied to the semi-natural grasslands of Malga Alparco (MA) and Bugacpuszta (BG). MA is an open mountain range and has no facility to record the stocking density of the grazing cattle; for this site the stocking rate was therefore estimated from farmer records and from measurements of the above-ground herbage productivity.

2.5. Annual budgets calculations

By convention, gross fluxes from the ecosystem to the atmosphere are added to the atmosphere budget. With this convention, a negative NEE value indicates a sink activity for the atmosphere. Adapting the definition of Chapin et al. (2002) to a managed grassland system, net biome productivity is calculated as

$$\mathrm{NBP} = \mathrm{NEE} - F_{\mathrm{import}} + F_{\mathrm{harvest}} + F_{CH_4} + F_{\mathrm{LW}} + F_{\mathrm{leach}} \tag{2}$$

where F_{harvest} is the C lost from the system through plant biomass export (mowing); F_{import} the flux of C entering the system through manure and/or slurry application; F_{CH_4} the C lost through CH_4 emissions by grazing cattle; F_{LW} the C lost from the system through animal body mass increase and milk production; F_{leach} is the C lost through dissolved organic/inorganic C leaching. In this study F_{leach} and F_{LW} were not determined and will be neglected for the calculation of NBP (see Allard et al., 2007).

The net GHG exchange (NGHGE) for each site was calculated by adding CH_4 and N_2O emissions to NEE budgets using the global warming potential of each of these gases at the 100-year time horizon (IPCC, 2001):

$$\mathrm{NGHGE} = \mathrm{NEE} + F_{CH_4}\mathrm{GWP}_{CH_4} + F_{N_2O}\mathrm{GWP}_{N_2O} \tag{3}$$

where $\mathrm{GWP}_{N_2O} = 127$, as 1 kg N_2O-N = 127 kg CO_2-C; $\mathrm{GWP}_{CH_4} = 8.36$, as 1 kg CH_4-C = 8.36 kg CO_2-C.

In order to account for: (i) the off-site CO_2 and CH_4 emissions resulting directly from the digestion by cattle of the forage harvests and (ii) the manure and slurry applications which add organic C to the site, an attributed net greenhouse gas balance (NGHGB) was then calculated as

$$\mathrm{NGHGB} = \mathrm{NGHGE} + F_{\mathrm{harvest}}(f_{\mathrm{digest}} + f_{CH_4}\mathrm{GWP}_{CH_4}) \tag{4}$$

where f_{digest} is the fraction of the ingested C that is digestible and hence will be respired by ruminants and f_{CH_4} the fraction of the ingested C emitted as CH_4 from enteric fermentation. A fixed value for f_{digest} (0.65) was taken from Thornley (1998). f_{CH_4} was calculated as the ratio of the measured CH_4 emissions to the annual C intake.

The annual C intake (F_{intake}) by grazing cattle was estimated from the mean annual stocking density which can best be estimated in livestock units (SR_{LU}, livestock unit ha^{-1}).

$$F_{intake} = k_{intake}\,Gd\,SR_{LU} \quad (5)$$

where k_{intake} is the daily intake rate per livestock unit and Gd is the number of grazing days per year. A fixed value for k_{intake} (4.8 kg C LU^{-1} day^{-1}) was taken from Thornley (1998).

3. Results

3.1. Seasonal patterns of net ecosystem exchange of CO_2

The seasonal patterns of the net CO_2 exchange at each of the nine sites broadly reflect their position in the European continental gradient. This gradient is mainly characterized by the magnitude of temperature differences between summer and winter at each site (Table 1), or more precisely by the duration of the growing season. As pasture growth usually stops below 5 °C (Parsons, 1988), the potential (temperature based) duration of the growing season can be defined by the number of days per year with an average air temperature above this threshold temperature. Sites with an oceanic climate (CA, BS and LE) had a long potential growing season (above 260 days) with an average 210 days of C sink activity over the two measurement years. Sites with a more continental climate (OEi, OEe, BG, LAi, and LAe) had a relatively short potential growing season (less than 250 days) and displayed, on average, a slightly shorter C sink activity (approximately 190 days each year). Finally, the high elevation MA site had both a short growing season (178 days) and a short C sink activity time period (158 days). However, no significant relationship was found between the duration of the potential growing season and the number of days with a net CO_2 uptake (data not shown).

The daily minimum in NEE (Fig. 1) provides a proxy of the gross primary productivity (GPP). All sites displayed a rapid increase in absolute value of mid-day NEE C uptake at the end of spring/beginning of summer, reaching a peak which ranged from -18 μmol CO_2 m^{-2} s^{-1} for a semi-natural grassland in a dry continental climate (BG site) to -35 μmol CO_2 m^{-2} s^{-1} for sown grass-clover mixtures in a wet continental site (OE site) (Fig. 1).

The time course of the mid-day NEE was also markedly affected by the cutting and grazing management. The absolute value of mid-day NEE was sharply reduced after most cutting events at CA, OEi, OEe, LV and LE sites. Moreover, the mid-day NEE was reduced after the start of grazing in LAi, LAe, CA and MA (Fig. 1).

The daily and seasonal time course of net CO_2 exchange in each site is shown in Fig. 2. The daily duration of CO_2 uptake is constrained by day length which varies according to a sine function over the course of the year. In spring and summer, periods of high CO_2 uptake during day time are associated with strong respiratory fluxes at night. Conversely, in winter the magnitude of the CO_2 exchanges is sharply reduced with some sites displaying a net source activity during day time. The high elevation site of MA displays a short but nevertheless active CO_2 uptake time period. A dry continental semi-natural grassland site as BG displays both a short time period of sink activity and a low magnitude of this activity. In contrast, highly productive managed grasslands (e.g. OE, CA, BS, LE) display a strong seasonal peak of uptake and tend to have a prolonged sink activity during the growing season. Net CO_2 uptake is interrupted in some sites by the summer water deficit, which was more marked during summer 2003, compared to summer 2004, in OE, LA, BG and possibly MA (Fig. 2).

3.2. Carbon budgets

All grassland sites showed a negative annual NEE budget (Table 4) with large between site variability. The strongest annual CO_2 sink activity was recorded in MA (-464 g C m^{-2} $year^{-1}$ in year 1) while LAe had the lowest (-49 g C m^{-2} $year^{-1}$ in year 2) CO_2 sink activity. The between year variability was strong for some sites, especially under grazing management. For example, the BG site had an increased CO_2 sink activity of +112 g C m^{-2} $year^{-1}$ in year 2 compared to year 1. At MA, the sink activity was reduced by 45% in the second compared to the first year of measurement. The average NEE for all grassland sites was -247 ± 67 g C m^{-2} $year^{-1}$ (mean ± confidence interval at $p > 0.95$, Fig. 4). The crop-grass site of the network (LV) displayed a NEE budget of -31 and -373 g C m^{-2} $year^{-1}$ in years 1 and 2, respectively.

The NBP of a grassland plot can be calculated from NEE by taking into account imports and exports of organic carbon and losses of carbon as methane (see Eq. (2)). The contribution of F_{CH_4} (expressed in CH_4-C) to NBP was small (Table 4). In contrast, the balance between imports (F_{import}) and exports ($F_{harvest}$) of organic C created a large departure of NBP from NEE. The average C storage (net biome productivity, NBP) of all sites was estimated at -104 ± 73 g C m^{-2} $year^{-1}$ (Fig. 4), that is 43% of the atmospheric CO_2 sink. The confidence interval at $p > 0.95$ indicated that the NBP was significantly different from zero among sites and years (Fig. 3). Moreover, a sign test with means per site ($n = 9$, $p < 0.05$) showed that NBP was significantly different from zero.

Sites which were partly or only cut (BS, CA, LE, OEi and OEe) displayed C exports in the range of 220–476 g C m^{-2} $year^{-1}$ (Table 4). C import caused by manure or slurry application did not compensate for these exports, except for the crop-rotation LV site that received a large amount (1400 g C m^{-2} $year^{-1}$) of horse manure, on average over the two experimental years.

3.3. GHG budgets in CO_2-C equivalents

The field scale GHG budgets were calculated as the sum of NEE, CH_4 and N_2O fluxes, the two latter being corrected

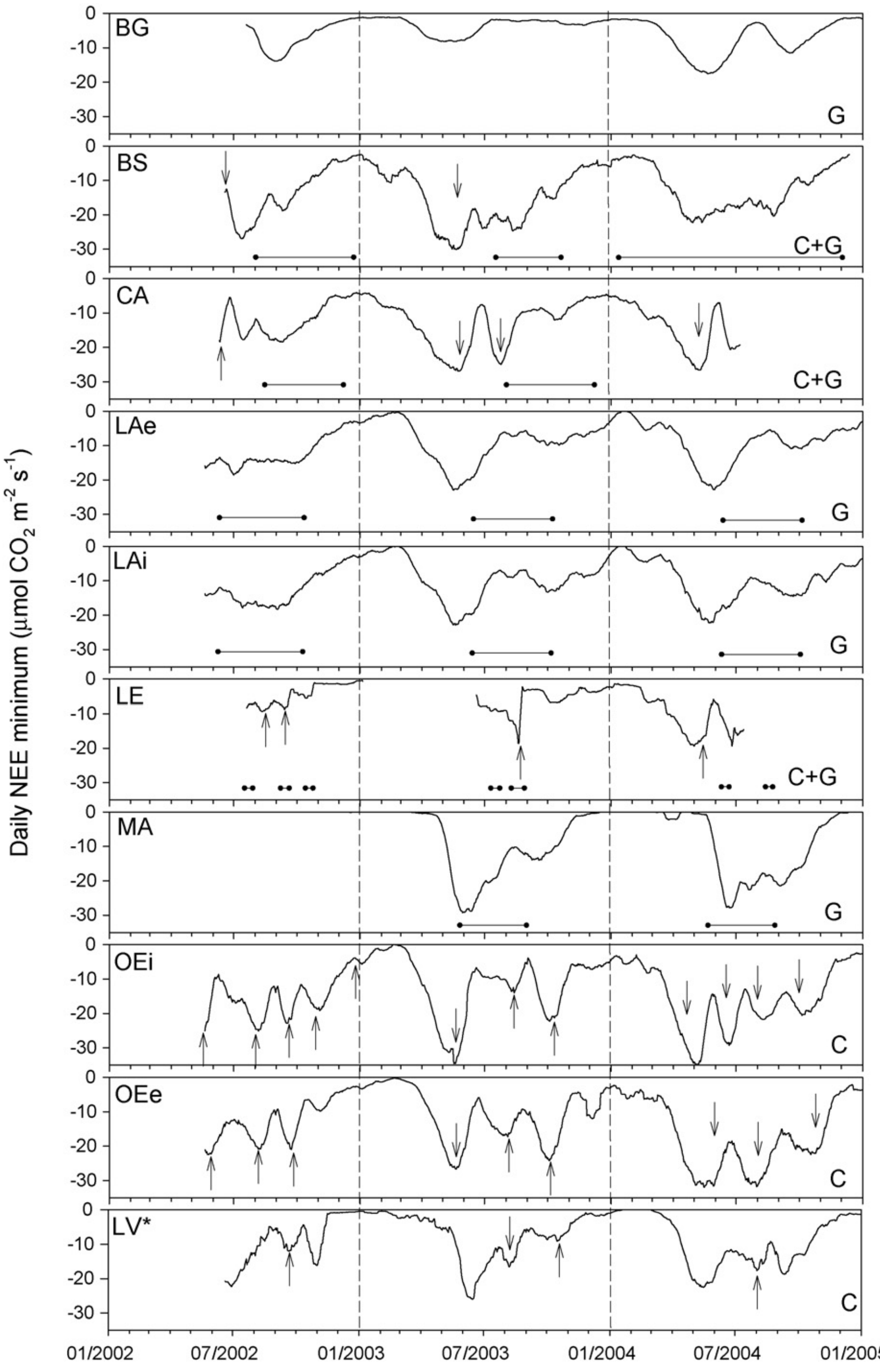

Fig. 1. Daily minimum NEE at the GREENGRASS sites over the experimental period. The management options at each site are shown: grazing (G), cutting (C), both (C + G). Grazing events are depicted by a horizontal line and cutting events are indicated by a black arrow.

by their global warming potential at the 100-year time horizon. On average over all sites, N_2O fluxes reached 13 g CO_2-C equiv. m^{-2} $year^{-1}$ (Table 4). Site variability was large ranging from sites with high N_2O emissions (LE, 87.2 g CO_2-C equiv. m^{-2} $year^{-1}$ in year 2) to sites that displayed negative N_2O fluxes (OEe both years) (Table 4). Significant negative fluxes were measured at Oensingen, especially on the unfertilized field (OEe), simultaneously with sub-ambient N_2O concentrations in the soil, indicative of a consumption process which is active in dry as well as in wet conditions (Flechard et al., 2005).

CH_4 emissions varied with the annual animal stocking density which can best be estimated in livestock units (SR_{LU}) (Table 3) and which displayed an approximate 10-fold variation from 0.16 (BG) to 1.3 (LAi) LU $year^{-1}$. The methane emission rate (k_{CH_4}) per unit liveweight and per year was also markedly different between animal types. This rate was comprised between 0.33 and

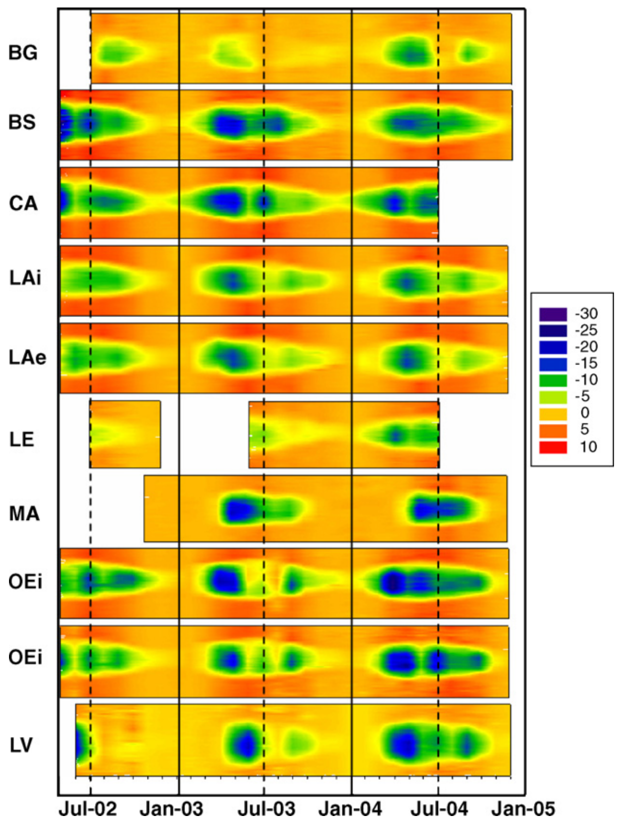

Fig. 2. Mean diurnal NEE variation (in μmol CO_2 m^{-2} s^{-1}) at the GREENGRASS sites along the experimental period. In each horizontal bar, diurnal changes in NEE are plotted using a colour scale centered on noon (solar time).

0.45 g CH_4 kg^{-1} LW $year^{-1}$ for heifers and bulls (LAi, LAe and CA) and reached 0.68–0.97 g CH_4 kg^{-1} LW $year^{-1}$ for lactating cows (LE) (Table 3). On average, over the four sites managed by grazing for which CH_4 emissions were directly measured (CA, LE, LAi and LAe) CH_4 emissions reached 86 g CO_2-C equiv. m^{-2} $year^{-1}$ (Table 4), i.e. a 35% trade-off (range 13–95%) of the NEE in these sites. For the remaining grazed grassland sites (BG, BS and MA), CH_4 emissions were calculated by using off-site k_{CH_4} values (see Section 2) and were estimated to induce a 9% trade-off (range 5–14%) of NEE. Therefore, even a large error in k_{CH_4} would not lead to a large change in the net greenhouse gas balance of these sites.

Averaging of all grassland sites, the net GHG exchange (NGHGE) reached -202 ± 76 g CO_2-C equiv. m^{-2} $year^{-1}$ (Fig. 4). The GHG budget of the crop-grass rotation site (LV) also exhibited a sink activity of -195 g CO_2 C m^{-2} $year^{-1}$ on average over the 2 years (Table 4).

When the attributed GHG balance (NGHGB) was considered through the accounting of off-site CO_2 and CH_4 emissions resulting from the digestive use by herbivores of the cut herbage, the grasslands of the networks were, on average, a GHG sink of -85 ± 77 g CO_2-C equiv. m^{-2} $year^{-1}$. The confidence interval at $p > 0.95$ indicated that the NGHGB was significantly different from zero among sites and years (Fig. 3). Nevertheless, a sign test with means per site ($n = 9$) indicated that NGHGB was not significantly different from zero at $p < 0.05$.

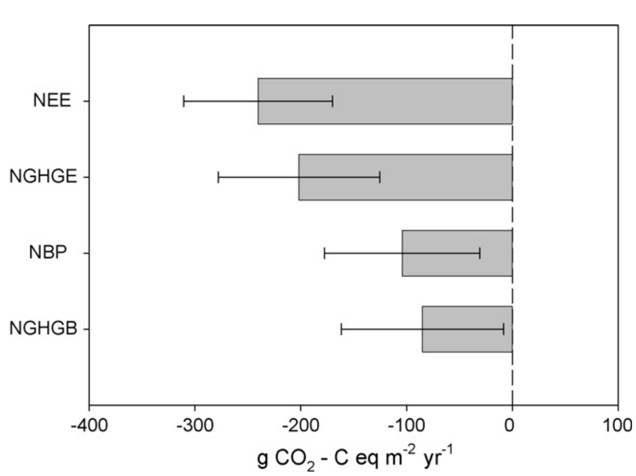

Fig. 3. Average NEE, NBP, GHG budget (NGHGE) and attributed GHG budget (NGHGB) over the GREENGRASS grassland sites (excluding the grass crop rotation site, LV). Results are means (± confidence interval at $p > 0.95$) of nine sites and of 2 years per site.

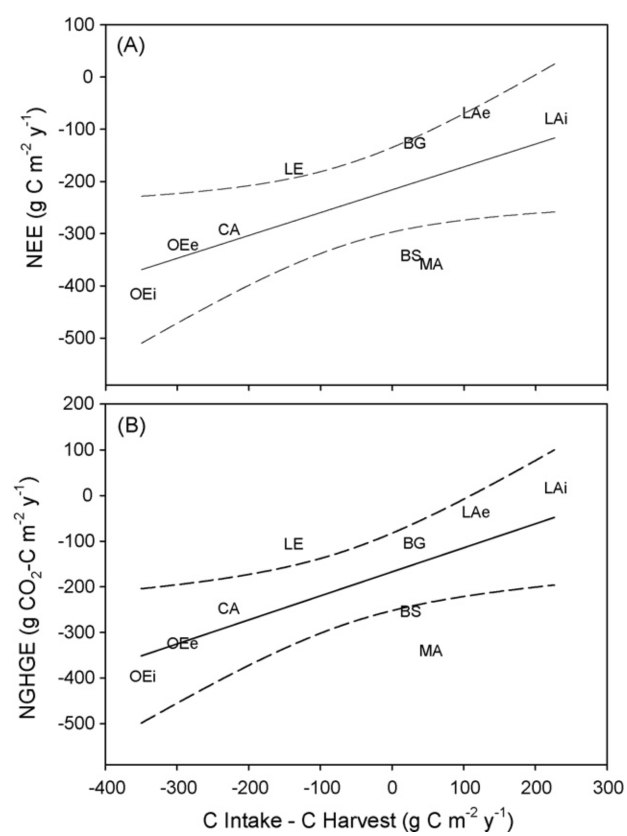

Fig. 4. Relationships between the NEE (A), the net greenhouse gas exchange (NGHGE) (B) and the difference between annual carbon intake through grazing and annual carbon export through cutting. Regression lines plotted are: (A) $n = 9$, $r^2 = 0.43$, $p < 0.05$, $y = -208 + 0.45x$; (B) $n = 9$, $r^2 = 0.48$, $p < 0.05$, $y = -163 + 0.53x$.

Table 3
Grazing mode and duration, average animal stocking rate in heads (SR) and in livestock units (SR_{LU}), mean animal liveweight (W), methane emission rate per unit liveweight (k_{CH_4}) and annual CH_4 emissions from enteric fermentation (E_{CH_4}) at the grazed sites in the GREENGRASS network[a]

Site[b]	Year	Grazing mode	Animal type	Grazing duration (day year^{-1})	SR (head ha^{-1} year^{-1})	W (kg LW head^{-1})	SR_{LU} (LU ha^{-1} year^{-1})	F_{intake}[c] (g C m^{-2} year^{-1})	k_{CH_4} (g CH_4 kg^{-1} LW year^{-1})	E_{CH_4} (kg CH_4 ha^{-1} year^{-1})
BG	1	Continuous	Heifers	81	0.19	380	0.12	21	*0.43*	*11.3*
	2	Continuous	Heifers	184	0.34	380	0.22	38	*0.43*	*20.3*
BS	1	Continuous	Heifers/sheep	150	2.5	200	0.83	146	*0.43*	*78.5*
	2	Continuous	Heifers/sheep	250	2.1	200	0.70	123	*0.43*	*65.9*
CA	1	Continuous	Bulls	117	0.7	713	0.83	146	0.37	67.4
	2	Continuous	Bulls	137	0.8	609	0.81	142	0.29	51.6
LAi	1	Continuous	Heifers	157	1.5	505	1.26	221	0.48	132.7
	2	Continuous	Heifers	152	1.6	496	1.32	232	0.4	115.9
LAe	1	Continuous	Heifers	157	0.8	496	0.66	116	0.45	65.2
	2	Continuous	Heifers	152	0.8	487	0.65	114	0.42	59.7
LE	1	Rotational	Dairy cows	3 × 5	0.7	560	0.65	115	0.97	138.8
	2	Rotational	Dairy cows	2 × 5	0.4	560	0.37	65	0.68	55.6
MA	1	Continuous	Heifers and bulls	100	*0.3*	600	0.3	53	*0.43*	*28.3*
	2	Continuous	Heifers and bulls	100	*0.3*	600	0.3	53	*0.43*	*28.3*

[a] Values in italics were estimated (see Section 2).
[b] Abbreviations see Table 1.
[c] The annual carbon intake (F_{intake}) was estimated according to Eq. (5). A livestock unit has a standard liveweight of 600 kg head^{-1}.

Table 4
Annual greenhouse gas fluxes[a] and their budget in CO_2-C equivalents[b] at sites of the GREENGRASS network[c]

Site[d]	Time period	Year	NEE (g C m^{-2} $year^{-1}$)	F_{CH_4} (g C m^{-2} $year^{-1}$)	$F_{harvest}$ (g C m^{-2} $year^{-1}$)	F_{import} (g C m^{-2} $year^{-1}$)	NBP (g C m^{-2} $year^{-1}$)	E_{N_2O} (mg N_2O-N m^{-2} $year^{-1}$)	EC_{N_2O} (g CO_2-C eq. m^{-2} $year^{-1}$)	EC_{CH_4} (g CO_2-C eq. m^{-2} $year^{-1}$)	NGHGE (g CO_2-C eq. m^{-2} $year^{-1}$)	NGHGB (g CO_2-C eq. m^{-2} $year^{-1}$)
BG	July 2002–June 2004	1	−13	0.8	0	0	−12	102	13.0	7	7	7
		2	−125	1.5	0	0	−124	86	10.9	13	−101	−101
BS	June 2002–May 2004	1	−384	5.9	220	−3	−161	230	29.2	49	−305	−88
		2	−302	4.9	0	−3	−300	57	7.2	41	−254	−257
CA	June 2002–May 2004	1	−372	5.1	271	0	−96	4.7	0.6	42	−329	−74
		2	−214	3.9	476	0	266	113	14.4	32	−168	251
LAi	May 2002–April 2004	1	−50	9.1	0	0	−41	66	8.4	79	37.4	37
		2	−112	9	0	0	−103	78	9.9	75	−27	−27
LAe	May 2002–April 2004	1	−91	5.4	0	0	−86	19	2.4	43	−46	−46
		2	−49	4.6	0	0	−44	17	2.2	38	−9	−9
LE	June 2002–May 2004	1	n.d.[e]	10.4	237	−104	n.d.	687	87.2	87	n.d.	n.d.
		2	−177	4.2	220	−80	−33	279	35.4	35	−107	151
MA	December 2002–November 2004	1	−464	2.1	0	0	−462	0.8	0.1	18	−446	−446
		2	−255	2.1	0	0	−253	1.6	0.2	18	−237	−237
OEi	May 2002–April 2004	1	−419	0	460	−106	−65	198	25.2	0	−394	−95
		2	−414	0	240	−29	−203	104	13.2	0	−401	−245
OEe	May 2002–April 2004	1	−352	0	380	0	28	−25	−3.2	0	−355	−108
		2	−293	0	210	0	−83	−23	−2.9	0	−296	−159
LV[f]	June 2002–May 2004	1	−31	0	406	−1450	−1075	80	10.1	0	−21	–
		2	−373	0	259	−1365	−1479	30	3.8	0	−369	–

[a] NEE, net ecosystem exchange; F_{CH_4}, carbon lost as CH_4; $F_{harvest}$, carbon exported by hay or silage cuts; F_{import}, C imported by manure and slurry applications; NBP, net biome productivity; E_{N_2O}, N_2O emission.

[b] EC_{N_2O}, EC_{CH_4}, N_2O and CH_4 emissions in CO_2-C equivalents; NGHGE, net greenhouse gas exchange in CO_2-C equivalents; NGHGB, attributed net GHG balance calculated by subtracting from GHGE off-site emissions of CO_2 and CH_4 (in CO_2-C equivalents) resulting from the digestion by cattle of exported forage and by adding organic carbon imports (see Section 2).

[c] Values in italic were estimated (see Section 2).

[d] Abbreviations see Table 1.

[e] n.d., not determined.

[f] The LV site is a crop-grass rotation.

4. Discussion

When assessing the impact of land use and land use change on greenhouse gas emissions, it is important to consider the impacts on all greenhouse gases (Robertson et al., 2000; Smith et al., 2001). This study provides for the first time a simultaneous accounting of the net exchanges of CO_2, N_2O and CH_4 at a range of European grassland sites covering a major climate gradient and including a variety of grassland types and managements. This opens the possibility to calculate the budget per unit land area of GHG exchanges in CO_2-C equivalents.

4.1. Grassland site network

The selection of sites is crucial for any network and may induce a bias (Korner, 2003). Our sites included an important climate gradient (continentality) and diverse grassland types and managements. Climate conditions induce a grassland production potential. Most grasslands are used according to this potential and, therefore, the total C use ($F_{harvest} + F_{intake}$) was significantly correlated ($p < 0.05$, data not shown) with the number of days above 5 °C. Moreover, in the site network, the herbage production (C use by grazing and cutting) tended to be correlated ($p < 0.10$) with the total N input from fertilizers, which was also correlated with the climate. Therefore, climate and management effect cannot be easily separated.

4.2. Net ecosystem exchange of CO_2

The annual NEE values indicated a sink activity for atmospheric CO_2 at all grassland sites as well as at the crop-grass rotation (LV) site (Table 4). The magnitude of the mean sink activity (-240 ± 70 g C m^{-2} $year^{-1}$) from the grassland covers (Fig. 3) is in the same range as found for forest sites in Europe (Valentini et al., 2000), with highest NEE values close to those found for coniferous (evergreen) forests.

Janssens et al. (2003) concluded that European grasslands may constitute a net C sink (-600 ± 800 g C m^{-2} $year^{-1}$), although the uncertainty surrounding this estimate was larger than the sink itself. Our findings for the grassland NEE show a significant atmospheric ($n = 9$, sign test, $p < 0.01$) sink activity for CO_2, the magnitude of which is nevertheless lower than the mean estimate by Janssens et al. (2003) and with a coefficient of variation of 62% among sites. With an extended data set covering 20 European grassland sites, Gilmanov et al. (2007) show that four sites only have the potential to be C sources in some years, two of them during drought events and two of them with a significant peat horizon. These findings for European grasslands confirm earlier estimates for North America (Follett, 2001) that these ecosystems predominantly act as a sink for atmospheric CO_2.

However, as shown by Ciais et al. (2005), the magnitude of the CO_2 sink activity is sensitive to heat and drought, which affect both gross photosynthesis and total ecosystem respiration. 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. Compared to 2002, the summer 2003 heat wave reduced the magnitude of NEE in some continental sites (BG, OE and LA, Figs. 1 and 2) which experienced higher summer temperatures (+5 to 6 °C for the mean summer temperature in OE and LA) and lower rainfall. However, at an annual time scale, there is no evidence of a NEE decline in 2003, possibly because of a warm spring prior to the development of the excess heat and drought.

The duration of the CO_2 uptake period is strongly constrained by the temperature dependency of plant growth. For example MA, a mountain site characterized by a particularly short growing season period (Table 1) exhibits a short duration NEE activity centered on the summer months (Fig. 2). In contrast, Atlantic sites (CA and BS) display a long growing season with active CO_2 exchange (Fig. 2). Nevertheless, climatic drivers fail to be good predictors of the between sites variability of annual NEE. For example, despite its short growing season, the MA site displays a stronger mean sink activity than that of the Atlantic site of CA (Table 4).

NEE is a small difference between two gross fluxes of opposite direction (gross primary productivity, GPP and total ecosystem respiration, TER) which are each driven by temperature and precipitation (Gilmanov et al., 2007). In contrast to forest ecosystems, grassland canopies are subjected to frequent defoliation through cutting and grazing events which reduce both leaf area index and above-ground herbage mass, thereby affecting CO_2 uptake and release by the vegetation. Cutting induces an abrupt decline in mid-day NEE, while the impacts of grazing are more gradual since only part of the available herbage is defoliated each day by herbivores (Fig. 1).

The role of grazing versus cutting for the annual NEE can be further illustrated by plotting NEE against the balance between carbon intake during grazing and carbon export through cutting (Fig. 4A). NEE is positively correlated ($p < 0.05$) with this balance (Fig. 4), since the CO_2 exchanges measured by the masts include herbivore's respiration in grazed systems but not in cut systems. In cuts grasslands a large part of the atmospheric CO_2 sink is stored in a labile forage pool which will be digested off-site by herbivores, thereby releasing to the atmosphere the digestible carbon fraction. The role of horizontal C fluxes induced by the agricultural management of grasslands can be better understood by calculating the net biome productivity (NBP).

4.3. Net biome productivity

In addition to NEE, the full budgeting equation for NBP in grasslands (Eq. (2)) includes four components: (i) disturbance leading to organic C exports (mowing) and imports (manure and slurry application), (ii) CH_4 emissions by herbivores, a gaseous CH_4-C loss, (iii) dissolved organic/inorganic (DOC/DIC) carbon losses to water and (iv) carbon exports in animal products (milk and meat production). Other presumably minor components, such as the emission of volatile organic C compounds (Kesselmeier et al., 2002), were not considered in this calculation.

In practice, not all components of the budget were measured in the site network and a simplified equation had to be used to calculate NBP by neglecting DOC/DIC losses as well as C exports in milk and meat products. Siemens and Janssens (2003) have estimated at the European scale the average DOC/DIC loss at 11 ± 8 g m^{-2} year^{-1}. Assuming a value at the upper range of this estimate, would reduce the grassland NBP by 20%, without changing the conclusion ($p < 0.05$, sign test) of a negative mean NBP and hence of an average carbon storage by the grassland ecosystems studied.

In comparison, to potential DOC/DIC losses the role of organic C exports in meat and milk products (F_{LW}) is small. For example, with the continuously grazed LAe and LAi sites, C accumulation in cattle liveweight gain accounted for only 1.6% of the NBP (Allard et al., 2007).

According to Eq. (2), NBP is partly uncoupled from NEE. This is illustrated by the 43% offset between NBP and NEE on average of the site network (Fig. 3). Indeed, the NBP was not significantly correlated with NEE and there was also no significant correlation between NBP and the climate factors listed in Table 1 (data not shown). With experimental grassland ecosystems, Verburg et al. (2004) proposed two estimates of C budget based on the fate of the exported C, that was considered alternatively to be potentially fully retained in the system (NBP = NEE) or fully lost (NBP = NEE − exports). In grasslands usually managed through regular prescribed burning like the tallgrass prairie (Suyker and Verma, 2001), Suyker et al. (2003) observed over three consecutive years an average C sink activity of -148 g C m^{-2} year^{-1}. However, when the C loss caused by the prescribed burns was accounted for, the net CO_2 exchange turned to a source of 62 g C m^{-2} year^{-1}.

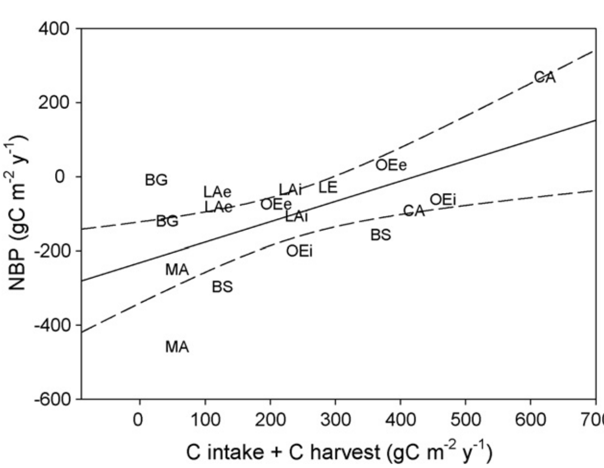

Fig. 5. Relationships between net biome productivity and C content of the total herbage used by grazing and cutting. Regression line plotted is: $n = 17$, $r^2 = 0.37$, $p < 0.01$, $y = -(231 \pm 53) + (0.55 \pm 0.19)x$.

In grazed only systems (not supplied with manures), NEE is indeed a good proxy of net C storage. Plant biomass is digested on-site by the herbivore and this process contributes to the total ecosystem respiration that can be analyzed from eddy covariance data (see Gilmanov et al., 2007). By contrast, in cut grasslands, biomass is exported off-site and neither this carbon export, nor the import of carbon from organic fertilizers, is detected by the atmospheric budget. Therefore, accounting for exports and imports of organic carbon is essential to compare cut and grazed grasslands in terms of their net carbon storage (NBP) (Yazaki et al., 2004). Numerous studies should be re-analyzed following this precept (Novick et al., 2004; Rogiers et al., 2005).

Means per site and per year indicate that NBP was positively correlated ($n = 17$, $r^2 = 0.37$, $p < 0.01$) to total C use (by grazing and cutting) (Fig. 5). Moreover, a multiple regression (Eq. (6)) including the year-to-year variability indicates that NBP is negatively correlated to N supply and positively correlated with total C use:

$$\begin{aligned} \text{NBP} = & -(200 \pm 50) - (0.68 \pm 0.33)\ \text{N supply} \\ & + (0.77 \pm 0.20)(F_{\text{intake}} + F_{\text{harvest}}), \\ & n = 17,\quad r^2 = 0.51,\quad p < 0.01 \end{aligned} \tag{6}$$

This relationship shows that net carbon storage is stimulated by N fertilizer supply and is reduced by the total C use through cutting and grazing. According to this equation, in the absence of both N supply and herbage use, NBP is negative and, therefore, unmanaged grasslands are also predicted to store carbon.

Grassland management methods that increase forage production such as N fertilization have been shown to have the potential to increase soil C stocks (Conant et al., 2001; Rees et al., 2005). First, N supply increases net primary productivity in N limited ecosystems (Chapin et al., 2002). Second, N supply may increase the proportion of C that remains in the ecosystem. Carbon storage can be sustained in the long-term only if nitrogen is added to the ecosystem (e.g. through N deposition, N_2 fixation, N fertilizer supply) (Hungate et al., 2003). Soil C losses tend also to increase when soil microbes are nitrogen limited (Fontaine et al., 2004) and a theoretical model of the complex relationship between C input and soil C sequestration has recently shown that N supply is a key to sustained C storage in SOM (Fontaine and Barot, 2005). Moreover, a moderate increase in N supply to permanent grasslands has indeed been shown in long-term surveys to increase grassland top soil organic C stocks at a rate of ca. 20 g C m^{-2} year^{-1} (Soussana et al., 2004). There are however exceptions with nutrient poor grasslands developed on organic soils, which may respond to fertilizer N supply by losing carbon (Soussana et al., 2004).

Disturbance by grazing and cutting may reduce C storage in grasslands through a decline in net primary productivity caused by a reduction in leaf area index (Parsons et al., 1983). Moreover, a large share of the NPP is exported by cutting in mown grasslands, amounting to 21 and 48% of the net canopy photosynthesis in sown grasslands at low and high N supply, respectively (Casella and Soussana, 1997). Grazing also strongly reduces the share of the NPP that can accumulate in the ecosystem. Under intensive grazing, up to 60% of the above-ground dry matter production is ingested by domestic herbivores (Lemaire and Chapman, 1996). The non-digestible carbon (25–40% of the intake according to the digestibility of the grazed herbage) is returned to the pasture in excreta (mainly as faeces). In most European husbandry systems, the herbage digestibility tends to be maximised by agricultural practices such as frequent grazing and use of highly digestible forage cultivars. Consequently, the share of the net primary production that is returned to the soil declines when the grazing pressure increases, as only a small fraction of the ingested carbon is returned to the soil. This explains why both F_{intake} and $F_{harvest}$ tend to reduce NBP (Fig. 5 and Eq. (6)). Again, these conclusions are only valid within a gradient of managed grassland that does not cover the full range of grazed ecosystems. In particular, natural grazing ecosystems or rangelands may exhibit a compensatory response of NPP to moderate grazing (McNaughton, 1993).

In contrast to a number of assumptions in the literature, our results do not confirm the concept of carbon sink saturation (Watson et al., 2000). Almost all models of soil organic matter turnover assume that, in the absence of changes in environmental factors and in land use and land management, an equilibrium value will be reached for all soil organic C (SOC) pools (e.g. Hénin and Dupuis, 1945; Freibauer et al., 2004). In the site network, permanent semi-natural grasslands displayed a large NBP (e.g. MA, LAe, LAi,), while newly sown grass-clover mixtures (CA and OEe) displayed a net loss of carbon 1 year out of two, on average (Table 3). This questions the conventional wisdom that unmanaged systems tend to be at equilibrium carbon wise and that sown grasslands store carbon. More grassland sites clearly need to be investigated in a comparable way before being able to conclude, nevertheless Fig. 5 and Eq. (6) provide a clear indication that extensively managed (but N rich) grasslands may store more carbon than highly intensive grasslands.

4.4. Greenhouse gas balance

Budgeting equations can be extended to include emissions of non-CO_2 radiatively active trace gases and calculate a net exchange rate in CO_2-C equivalents (Eq. (3)). When converted in C equivalents, N_2O emissions reached on average of the nine sites, 6% (14 g equiv. CO_2-C m^{-2} $year^{-1}$) of the NEE. However, some sites with high N availability displayed very high N_2O emissions (LE, 87.2 g CO_2-C equiv. m^{-2} $year^{-1}$ in year 1) (Table 4). A detailed analysis of the sources of variability in the N_2O emissions and a calculation of the corresponding emission factors is provided by Flechard et al. (2007) and this study shows the role of temperature and of soil water pore filling for the spatial and temporal variability of emissions. A net uptake of N_2O occurred in one site (OEe) (Flechard et al., 2005).

CH_4 emissions varied with the annual animal stocking density which can best be estimated in livestock units (SR_{LU}) (Table 3) and which displayed a ca. 10-fold variation from 0.16 (BG) to 1.3 (LAi) LU $year^{-1}$. The stocking density was low in extensively managed semi-natural grasslands (e.g. BG, MA and LAe) and high in the intensive grazing management which was applied to the different grassland types (semi-natural, LAi, intensive permanent, BS and LE, sown mixture, CA). The methane emission rate (k_{CH_4}) per unit liveweight and per year was also markedly different between animal types. However, the rate for each animal type varied between 0.33 and 0.45 g CH_4 kg^{-1} LW $year^{-1}$ for non-lactating cattle (heifers and bulls, LAi, LAe and CA) and reached 0.68–0.97 g CH_4 kg^{-1} LW $year^{-1}$ for lactating cows (LE) (Table 3). The corresponding emission factors for methane emissions at grazing appear to be higher than previous estimates (IPCC, 2001). On average, of the grazed only sites, CH_4 emissions reached 54 g CO_2-C equiv. m^{-2} $year^{-1}$ (Table 4), i.e. a 25% trade-off of the NEE in these sites.

On average of all sites and for the 2 years, the net GHG exchange (NGHGE) calculated in CO_2-C equivalents was highly correlated with NEE ($r^2 = 0.97$), with N_2O and CH_4 emissions resulting in a 19% trade-off of the NEE (Fig. 3). Since this trade-off is relatively small, the NGHGE was, on average, a significant sink ($n = 9$, $p < 0.05$, sign test). Nevertheless, in two grazed only sites (LAi and BG) the net emissions of N_2O and CH_4 were greater than NEE during the first year leading to a net source of GHG (Table 4). Being highly correlated with NEE, the NGHGE was also significantly correlated ($p < 0.05$) with the difference between the C harvest and the C intake fluxes (Fig. 4). This indicates that compared to cutting, grazing reduces the on-site sink activity for greenhouse gases. However, this calculation does not include the GHG emissions from the machinery that is used for harvesting hay and silage in cut grasslands.

The attributed net GHG balance (NGHGB) was calculated according to Eq. (4) considering that the cut and exported herbage will be digested off-site by herbivores, thereby leading to additional emissions of CO_2 (digestible fraction) and of methane (by enteric fermentation). The non-digestible fraction of the cut herbage is considered to be returned to the soil in this equation and is therefore not accounted as a C loss. With this accounting method, the GHG balance of the grassland plot and of its associated livestock is estimated. This calculation does not account, however, for harvest and post-harvest losses of herbage carbon and for the role of the diet quality which can affect the enteric fermentation (Vermorel, 1995).

The attributed net GHG balance (NGHGB) reached -85 ± 77 g CO_2-C equiv. m^{-2} $year^{-1}$. On average of the 2 years, seven sites had a negative, and two sites, a positive NGHGB and therefore the attributed net GHG balance was not significantly different from zero according to a sign test.

5. Conclusions

Despite the relatively small number of sites involved in this study, our results show that European grasslands are likely to act as large atmospheric CO_2 sinks. By contrast to forests, approximately half of the sink activity is stored in labile carbon pools (i.e. forage) that are digested off-site, usually within less than 1 year. When expressed in CO_2-C equivalents, N_2O and CH_4 emissions from grassland plots do not compensate the atmospheric CO_2 sink activity. Nevertheless, the off-site digestion by livestock of the harvested herbage leads to additional emissions of CO_2 and CH_4 which compensate the net GHG sink activity of the grassland plots. There is a clear need to investigate more grassland sites according to the methodology presented here in order to further reduce uncertainties and to test the hypothesis, supported by our results, that net carbon storage per unit ground area declines with C use by herbivores.

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