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The role of grazing management for the net biome productivity and greenhouse gas budget (CO_2, N_2O and CH_4) of semi-natural grassland

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Abstract

Over a 3-years period, the CO_2, N_2O and CH_4 fluxes exchanged with the atmosphere were studied in an upland semi-natural grassland site which was divided at the start of the experiment in two large paddocks continuously grazed by cattle. The soil at the site is an Andosol with high organic matter content. The intensively managed paddock was supplied with mineral N fertilizer and was grazed at a target sward height of 6 cm. The extensively managed paddock received no N fertilization and was stocked at half the stocking density of the intensive paddock. The net ecosystem exchange of CO_2 was continuously measured in each paddock using the eddy covariance technique. Nitrous oxide emissions were measured fortnightly in both paddocks using static chambers. Methane emissions by the grazing cattle were measured four times per year in each paddock using the SF_6 tracer method. Averaged across the 3 years, the two paddocks were net sinks of atmospheric CO_2 (97 and 75 g CO_2-C m^{-2} $year^{-1}$ in the intensive and extensive treatments, respectively). Taking into account the LW gain of the cattle and the C loss through methane emissions, the net C storage was estimated at 87 and 69 g C m^{-2} $year^{-1}$ in the intensive and extensive treatments, respectively. Emissions of nitrous oxide and methane reduced by 89 and 55% the atmospheric sink activity of the intensive and extensive treatments, respectively. The average greenhouse gas (GHG) balance across the 3 years was -10 and -31 g CO_2-C equivalents in the intensive and extensive treatments, respectively. However, the net biome productivity (NBP) and GHG sink activities increased over time in the intensive grazing treatment, whereas they declined after 1 year in the extensive treatment, possibly as a result of a reduced nitrogen status of the vegetation. It is concluded that the suppression of fertilizer N supply combined with a strong reduction in grazing pressure may not be able to increase in the short term the GHG sink per unit land area of managed grasslands.

Keywords: Carbon sequestration; Ecosystem respiration; Eddy covariance; Gross primary productivity

1. Introduction

The recent initiation of widespread eddy covariance networks (e.g. Fluxnet, Carboeurope-IP) has helped identify the magnitude of terrestrial sinks and sources of atmospheric carbon dioxide (CO_2) and their seasonal and interannual variability (Ciais et al., 2005). So far, the main focus has been on forest ecosystems (Valentini et al., 2000; Falge et al., 2002a,b). Much less attention has been given to the balance of CO_2 exchange with grasslands despite the fact that, on a global scale, this biome covers approximately the same area as forests (Adams et al., 1990) and may play a significant role in balancing the global C budget (Batjes, 1998; Scurlock and Hall, 1998).

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The GHG budget of European grasslands is still highly uncertain. Vleeshouwers and Verhagen (2002), further quoted by Janssens et al. (2003), applied a semi-empirical model of land use induced soil C disturbances to the European continent (as far east as the Urals) and inferred a C sink of 101 Tg C year^{-1} over grasslands (0.52 t C ha^{-1} year^{-1}) with uncertainties larger than the mean. Managed European grasslands are often fertilized to sustain productivity and thus emit nitrous oxyde (N_2O) to the atmosphere above the background level that is found in natural systems (Jarvis et al., 2001). European grasslands sustain an important number of domestic herbivores, 150 millions cows and 150 millions sheep, roughly 15% of the global animal population (FAO, 2004). Grazers impact the cycling of C and N within pastures *via* defoliation, excretal returns and mechanical disturbance. They emit CO_2 *via* their metabolic activity and methane (CH_4) through enteric fermentation (see Pinarès-Patino et al., 2007).

Most data concerning the C balance of grasslands have been obtained in the absence of grazers (Casella and Soussana, 1997; Frank and Dugas, 2001; Suyker and Verma, 2001; Verburg et al., 2004). Given the prevalence of grazing in these ecosystems, and its implications for N_2O and CH_4 emissions, there is a need to investigate both the C and GHG balance of grazed pastures under realistic conditions of agricultural management.

Simulations with the PaSim model of the effects of grazing management (Soussana et al., 2004) have suggested a decline with increasing stocking density in the GHG sink activity of managed grasslands. The model was first brought near to equilibrium assuming a management by cutting. Simulations showed that grazing management induced a C sink activity, which declined through time. Without N supply a new equilibrium of the simulated soil C stocks was reached within a few years, while with N fertilizer applications the C sink activity declined less rapidly (Soussana et al., 2004). From these simulation results, we hypothesized that an extensive grazing management would reduce emissions of methane and nitrous oxide, compared to an intensive grazing management, and would also enhance C storage at least during the first years after the application of the treatment.

In order to test this hypothesis, we conducted a 3 years study of the CO_2, N_2O and CH_4 fluxes exchanged with the atmosphere in upland semi-natural grassland, divided into two large paddocks that were continuously grazed by cattle. One paddock was managed intensively by supplying mineral N fertilizer and by grazing at a set sward surface height. The other paddock was managed extensively without N fertilizer supply and at half the stocking density of the intensive treatment. By avoiding the supply of manures and the removal of herbage by cuts, we suppressed organic C fluxes at the system boundaries that may confound the C balance of managed grasslands (Verburg et al., 2004). The soil at the site is an Andosol, derived from volcanic ash and characterized by large organic matter content.

We compared the annual C and greenhouse gas (GHG, in CO_2-C equivalents) budgets of the two treatments in order to assess the potential of different management regimes as mitigation options for managed grasslands.

2. Materials and methods

2.1. Site description

Measurements were made at an upland semi-natural grassland site (45°38′N, 2°44′E; 1040 m a.s.l.) near Laqueuille, Puy de Dôme, France during 2002–2004. The soil has developed on a basaltic bedrock at depths from 35 to 80 cm, and very occasionally surfacing. It is an Andosol (16% clay, 56% silt, 28% sand) with 18% organic matter and 11% C in the top 10 cm. The soil bulk density in this horizon is 640 kg m^{-3}. The mean annual precipitation reaches 1200 mm with monthly means above 80 mm (data averaged over 22 years for the periods 1972–1984 and 1995–2004; no data from 1985 to 1995). The mean annual temperature reaches 7 °C with a minimum of 0.7 °C in January and a maximum of 14.8 °C in August. Another characteristic of this upland site is the frequent occurrence of frost (115 days year^{-1}).

The experimental field (6.65 ha) was located on a flat plateau which was used at the start of the 20th century for arable crops but was converted to permanent grasslands at least 50 years before the start of the experiment and has been managed for the last 30 years by a combination of mowing and cattle grazing with applications of cattle slurry and manure.

2.1.1. Grassland management

In spring 2002, the experimental field was subdivided into two adjacent paddocks and both paddocks were then subjected between May and October to continuous grazing by Holstein-Friesian heifers. The smaller paddock (2.81 ha), hereafter referred to as “intensive”, was maintained during the growing season (May–October) to reach a target surface sward height of 6 cm by adjusting the number of animals weekly (put and take approach) (Hodgson, 1990). The second paddock, hereafter referred to as “extensive”, was maintained at half the stocking density of the intensive paddock. In spring 2002, before the start of the experiment, both paddocks received 80 kg N ha^{-1} year^{-1} as manure. After the start of the experiment, in years 2002, 2003 and 2004, respectively, the intensive paddock received 80, 174 and 176 kg N ha^{-1} year^{-1} in three split applications of ammonium nitrate. The extensive paddock received no fertilizer N after the start of the experiment.

2.2. Meteorological measurements

A weather station coupled with a data logger (Campbell Scientific Inc., Model CR-10X) was installed between the two paddocks (100 m apart from each eddy covariance mast, see below). It provided 30 min averaged values of global radiation (Rg), net radiation (Rn), incident photosynthetic photon flux density (PPFD), soil temperature (at soil surface and at 5–10–30–50 cm depths), soil water content (at 5–10–30–50 cm depths), air temperature, vapor pressure, wind speed, wind direction and precipitation.

2.3. Eddy covariance measurements

Both paddocks were equipped with an eddy covariance sensor array measuring at a 30 min time step fluxes of CO_2, sensible heat, latent heat and momentum from spring 2002. Each device included a fast response (20 Hz) sonic anemometer (Gill Instruments, Lymington, UK, Model Solent R3) and an open path CO_2–H_2O analyzer (LI-Cor Inc., Lincoln Nebraska, USA, Model LI-7500) installed at a height of 2 m in each paddock. Fluxes were adjusted for the variation in air density due to the transfer of water vapor (Webb et al., 1980) and were corrected for inadequate sensor frequency response after analysis of the cospectra calculated from this study. Quality checks of the data were done following Carboeurope-IP guidelines (Aubinet et al., 2000). An additional filtering criterion was applied based on friction velocity (u^*) values. A critical threshold of u^* ($u^* = 0.08$) below which CO_2 flux was strongly dependant on friction velocity was determined for our site using available night time data. Data with u^* below this threshold were systematically discarded. The footprint of the flux in both paddocks was evaluated as a function of wind speed, direction and stability according to criteria defined by the Carboeurope-IP project (M. Gockede, pers. commun.). The results of this study showed that, for both masts, the flux footprint was within the studied paddock during more than 90% of the time. Gaps and poor quality data were reconstructed according to the gap-filling strategy used in the Carboeurope-IP project (Reichstein et al., 2005).

2.4. Soil N_2O and CH_4 flux measurements

N_2O and CH_4 fluxes were measured using static chambers and gas chromatography (GC). Eight stainless steel frames were placed at random in each paddock in spring 2002. Each static chamber was made of PVC (diameter 60 cm, height 30 cm) and was equipped with thermocouples for measuring soil temperature at 5 cm depth. During measurement, the chambers were coupled to the frame using plastic clips that ensured they were airtight. N_2O and CH_4 concentrations in the chambers were measured over 3 h. Air samples were collected with two replicates per enclosure and then every hour ($t0$, $t1$, $t2$, $t3$) with a plastic syringe and injected immediately into an airtight plastic tube that had been previously evacuated. Samples were then stored until analysis by GC using a Varian 3400Cx analyser (Varian, USA). The measurements were made fortnightly and simultaneously in each paddock during the growing season (April–November). Annual N_2O emissions were calculated based on a linear interpolation of daily emission rates.

2.5. Ruminant CH_4 and CO_2 flux measurements

Eight measurements campaigns (2002: early June, mid July, late August and late September; 2003: late May, late June, late July and late September) were performed in both paddocks in order to quantify CH_4 and CO_2 emissions by cattle. Measurements were made on seven core animals per paddock which were selected on a live weight (LW) uniformity basis. CH_4 production was measured by the hexafluoride (SF6) tracer technique (Johnson et al., 1994) as described by Ulyatt et al. (1999). Full details about the methodology used in this particular experiment can be found in this issue (Pinarès-Patino et al., 2007).

Prior to each measurement campaign, the core animals were weighted so that CH_4 emissions could be expressed on a LW basis. Daily CH_4 emissions per kg of LW between measurement campaigns were calculated using linear interpolation. In addition, all animals present in each paddock were regularly weighted during the grazing period and daily LW gain was calculated using linear interpolation. Thus, annual CH_4 budget per unit ground area were calculated by multiplying the average CH_4 emission rate per kg of LW and the average animal LW per ground area.

2.6. Budgeting equation and sign convention for flux data

In this paper, all fluxes are presented following the sign convention used in micrometeorology. Fluxes (of CO_2, N_2O and CH_4) from the ecosystem to the atmosphere are positive while CO_2 fluxes from the atmosphere to the ecosystem are negative.

Eddy covariance measures net ecosystem exchange (NEE) of CO_2. NEE is the arithmetic sum of gross primary productivity (GPP) and total ecosystem respiration (TER):

$$\mathrm{NEE} = \mathrm{GPP} + \mathrm{TER} \tag{1}$$

To break down NEE into GPP and TER we used the algorithm developed by Reichstein et al. (2005). The day time respiration was calculated from measured night time values after correction for temperature dependency using a Q_{10} approach. Parameterization of the Q_{10} function was done using the relation between temperature and night time respiration values. TER includes respiratory losses of CO_2 by grazing ruminants (Rani), autotrophic respiration (Ra) and soil heterotrophic respiration (Rh). Rani was measured directly (see above) but no attempt was made to calculate Ra and Rh.

Adapting the definitions of Chapin et al. (2002) to managed grasslands, net ecosystem productivity (NEP) was calculated as

$$NEP = NEE + F_{leach} + F_{CH_4} \quad (2)$$

where F_{leach} is the C lost from the ecosystem through leaching of dissolved organic and inorganic C (DOC and DIC) and F_{CH_4} the flux of C from ruminants CH_4 emissions. In the present experiment, F_{leach} was not assessed.

Net biome productivity (NBP) explicitly takes into account C fluxes caused by disturbance (fire, harvest, C import in organic fertilizer). Therefore in managed grassland:

$$NBP = NEP + F_{harvest} - F_{import} + F_{LW} \quad (3)$$

where $F_{harvest}$ is the C exported through silage and hay cutting, F_{import} the C imported through manure or slurry application and F_{LW} is the C accumulated through animal LW gain. In the experiment reported here the grassland paddocks were managed by continuous cattle grazing without C imports or C exports. Hence

$$NBP = NEP + F_{LW} = NEE + F_{CH_4} + F_{leach} + F_{LW} \quad (4)$$

The total GHG budget was calculated by adding CH_4 and N_2O emissions to NEE values using the global warming potential (GWP) of each gas at the 100 years time horizon (IPCC, 2001). This budget does not include organic C fluxes (e.g. F_{LW}). Thus

$$GHG = NEE + F_{CH_4} \cdot k_{CH_4} + F_{N_2O} \cdot k_{N_2O} \quad (5)$$

where (IPCC, 2001):

$k_{N_2O} = 127$, since 1 kg N_2O-N = 127 kg CO_2-C

$k_{CH_4} = 8.36$, since 1 kg CH_4-C = 8.36 kg CO_2-C

Micrometeorological measurements started in May 2002 which was also the start of the 2002 grazing season. Therefore, the interannual variability and the annual budgets of greenhouse gases were assessed during over 12 months periods beginning on 1st May and ending on 30th April. Years 1, 2 and 3 refer to the periods May 2002–April 2003, May 2003–April 2004 and May 2004–April 2005, respectively.

2.7. Plant aboveground biomass and nitrogen content

During the vegetative growth period (May–October), aboveground herbage biomass was measured approximately every 3 weeks, in 8 quadrats (individual area = 1 m^2) per paddock. In each quadrat, herbage was cut with a handheld grass clipper at a height of approximately 4 cm. Herbage samples were then separated into dead and live biomass and were oven dried (50 °C to constant mass) and finely ground with a ball mill (Retsch, Germany). The samples were then analysed for N content with an automatic Kjeldahl analyzer (2300 Kjeltec Analyser unit, Moss, Denmark). The nitrogen nutrition index (NNI) was calculated according to Lemaire and Salette (1984) as

$$NNI = \frac{N\%}{4.8\,BM^{-0.32}}$$

where N% is the nitrogen content of the green herbage material (in %), and BM is the green herbage biomass in t DM ha^{-1}. The use of NNI allows the evaluation of the sward N nutrition status by taking into account the ontogenic decline of plant N concentration during sward regrowth and canopy closure (Lemaire and Gastal, 1997). The annual NNI value was calculated as an average of the measurements made over the growing season.

2.8. Data analysis

Herbage biomass and N content in each quadrat ($n = 16$) were analysed for paddock effect using a t-test at each

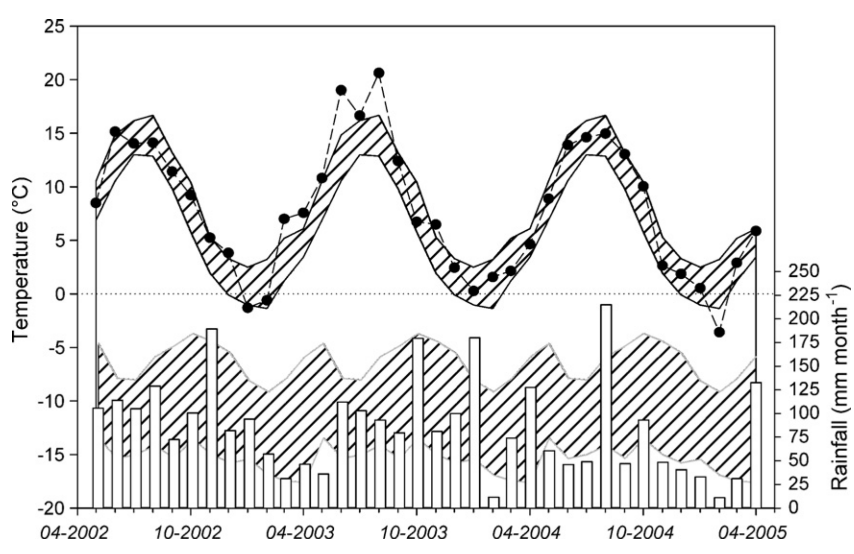

Fig. 1. Average monthly air temperature (full circles) and monthly total precipitation (open bars) at the Laqueuille site between Mai 2002 and Mai 2005. Air temperature during the measurement period is compared to the 20 years average ± S.D. at the same site (black hatched area) and rainfall data is compared to 20 years average ± S.D. (grey hatched area).

sampling date. CH_4 emissions per animal ($n = 14$) were also analysed for paddock effect using a *t*-test for each measurement period. The paddock was taken as a fixed factor for these tests which were performed using the Statgraphics (Magnugistics, USA) package.

3. Results

3.1. Climate

Monthly precipitation sums and mean air temperatures at the Laqueuille site over the course of the experiment are presented in Fig. 1. Yearly precipitation sums reached 1128, 1177 and 807 mm during years 1, 2 and 3, respectively. The 30% rainfall deficit recorded in year 3 was caused by a dry winter (Fig. 1). A major temperature anomaly was recorded in summer 2003 with monthly averages 5 and 6 °C warmer in June and August 2003, respectively, than the 20 years average. This corresponded to the unprecedented heat wave that occurred in Europe at that time. A cold event also occurred in February 2005 with air temperature averaging at −3.9 °C over that month (Fig. 1). This frost period was accompanied by a thick and long lasting snow cover.

3.2. Management

Grazing started on 21st May, 7th May and 6th May in years 1, 2 and 3, respectively. The time course of the instantaneous stocking rate in both paddocks is shown in Fig. 2. As described in Section 2, the stocking rate (SR) was adapted in the intensive paddock using the put and take approach to reach a target surface sward height of 6 cm. The instantaneous SR ranged from 1 to about 4 livestock unit (LU) ha^{-1} in the intensive paddock (and half of these values in the extensive paddock). The mean annual SR reached 0.9, 1 and 1.2 LU ha^{-1} $year^{-1}$ in years 1, 2 and 3, respectively, in the intensive paddock (half of these values in the extensive paddock) (Table 1). Animal LW at grazing season start was close to 400 kg in each year (Table 1). Average animal LW gain during the grazing season ranged between 84 and 137 kg LW $year^{-1}$ (Table 1).

Table 1
Mean annual animal stocking rate (SR), live weight (LW) at season start (LW_0), season end (LW_1) and LW gain in the intensive and extensive paddock

Year	Intensive	Extensive
Year 1		
SR (LU ha^{-1} $year^{-1}$)	0.9	0.5
LW_0 (kg $animal^{-1}$)	403	443
LW_1 (kg $animal^{-1}$)	509	527
LW gain (kg $animal^{-1}$)	**106**	**84**
Year 2		
SR (LU ha^{-1} $year^{-1}$)	1	0.5
LW_0 (kg $animal^{-1}$)	431	425
LW_1 (kg $animal^{-1}$)	532	523
LW gain (kg $animal^{-1}$)	**101**	**98**
Year 3		
SR (LU ha^{-1} $year^{-1}$)	1.2	0.6
LW_0 (kg $animal^{-1}$)	390	391
LW_1 (kg $animal^{-1}$)	490	528
LW gain (kg $animal^{-1}$)	**100**	**137**

3.3. CO_2 flux

The yearly time course of C accumulation in the ecosystem and release to the atmosphere had a typical seasonal pattern (Fig. 3). The NEE remained negative (atmospheric sink) during the active vegetation growth period (May–July), was close to zero during the end of summer and shifted to a source in autumn and winter. The grassland ecosystem then re-entered in a C stocking stage in the next spring (Fig. 3). Compared to years 2 and 3, the first year displayed lower levels of C storage in May and June especially in the intensive paddock. For this first year, the extensive paddock stored more C than the intensive paddock (−112 and −50 g C m^{-2} $year^{-1}$ for the extensive and intensive paddock, respectively). This trend was, however,

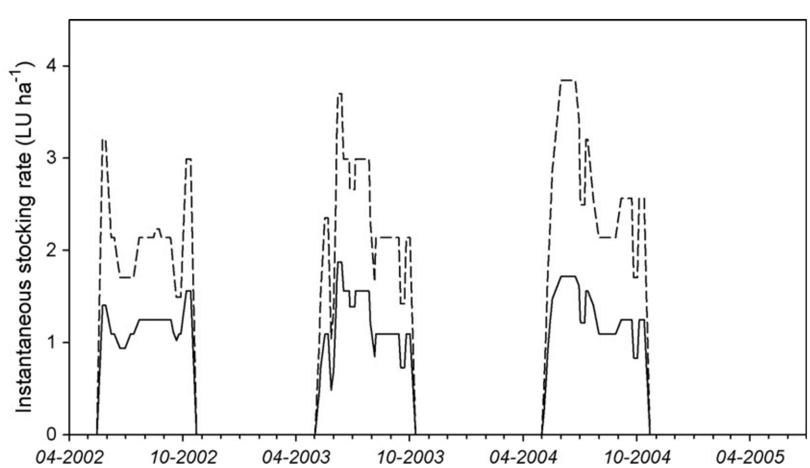

Fig. 2. Instantaneous stocking rate (LU ha^{-1}) applied in the intensive (dashed line) and extensive (full line) paddocks over the course of the experiment.

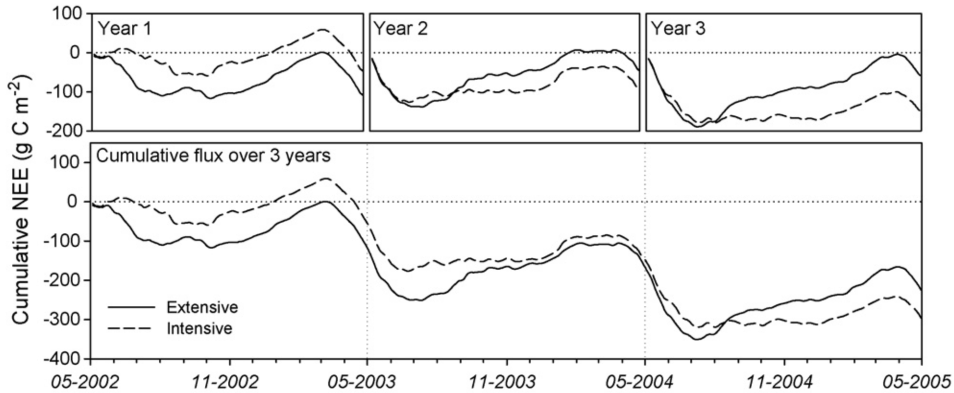

Fig. 3. Cumulative net ecosystem exchange of the extensive (solid line) and intensive (dotted line) paddocks. Cumulative fluxes are presented over 3 years and on a per-season basis.

reversed during the two following years (Fig. 3) with average annual C gains being greater by 42 and 91 g C m^{-2} in years 2 and 3, respectively, in the intensive compared to the extensive paddock (Fig. 3). On average of 3 years, both paddocks acted as a net C sink. The intensive paddock displayed an increasing C sink activity through time, while the opposite was found for the extensive paddock (Table 2 and Fig. 3).

The monthly NEE was broken down into gross primary productivity (GPP) and total ecosystem respiration (TER) (Fig. 4). The annual GPP varied between −1730 and −1400 g C m^{-2} $year^{-1}$. However, this flux was almost balanced by the TER and the NEE accounted for less than 10% of either of these gross fluxes. The time course of GPP contrasted periods of high (spring and summer) and low (winter) productivity. By comparison, the seasonal fluctuations of TER had a smaller magnitude. The GPP peaked earlier during the growing season than the TER (Fig. 4), which resulted in a strong atmospheric sink for CO_2 during spring. Each year, both GPP and TER were lower in absolute value in the extensive compared to the intensive paddock (Table 2). During year 3, stronger GPP and NEE sink activities were observed from August to November in the intensive compared to the extensive paddock (Figs. 3 and 4).

3.4. Aboveground biomass and nitrogen nutrition index

At the start of the growing season, the standing herbage mass was between 100 and 150 g DM m^{-2} in both paddocks (Fig. 5A). During spring and summer, more herbage accumulated in the extensive compared to the intensive paddock and the peak herbage aboveground mass was always greater in the extensive compared to the intensive paddock (Fig. 5A). Differences in green herbage biomass between paddocks were less marked over the course of the experiment, especially in autumn (Fig. 5B). The nitrogen nutrition index did not differ significantly among paddocks during the first 2 years of the experiment, but was significantly lower in the extensive compared to the intensive paddock in the third year (Fig. 6) which indicates an increasing N deficiency in the extensive paddock.

3.5. CH_4 fluxes from ruminant enteric fermentation

CH_4 emissions ranged from 0.36 to 0.52 g CH_4 day^{-1} kg^{-1} LW. There was no significant paddock effect for the CH_4 emissions per unit LW (Fig. 7). Per unit ground area, annual emissions of CH_4 from enteric fermentation were about two times higher in the intensive

Table 2
Annual budgets of C and GHG fluxes in the intensive (Int) and extensive (Ext) paddocks

Flux	Unit	Year 1		Year 2		Year 3		3 years average	
		Int	Ext	Int	Ext	Int	Ext	Int	Ext
GPP	g C m^{-2} $year^{-1}$	−1724	−1694	−1498	−1441	−1560	−1408	−1594	−1514
TER	g C m^{-2} $year^{-1}$	1674	1582	1407	1392	1405	1345	1495	1440
NEE	g C m^{-2} $year^{-1}$	**−50**	**−112**	**−91**	**−49**	**−155**	**−64**	**−99**	**−75**
F_{LW}	g C m^{-2} $year^{-1}$	1.6	0.7	1.6	0.8	2	1.3	1.7	0.9
F_{CH_4}	g CH_4-C m^{-2} $year^{-1}$	9.4	5.1	9	4.6	10.9	5.4	9.8	5.0
NBP	g C m^{-2} $year^{-1}$	**−39**	**−106**	**−80**	**−44**	**−142**	**−57**	**−87**	**−69**
CH_4	g CO_2-C m^{-2} $year^{-1}$	78.9	43	74.8	38.4	91.2	45.1	61.2	31.6
N_2O	g N_2O-N m^{-2} $year^{-1}$	0.066	0.019	0.078	0.017	0.03	0.003	0.058	0.013
	g CO_2-C m^{-2} $year^{-1}$	8.34	2.41	9.89	2.17	3.75	0.39	7.33	1.66
GHG	g CO_2-C m^{-2} $year^{-1}$	**37.2**	**−66.6**	**−6.3**	**−8.4**	**−60.1**	**−18.5**	**−9.7**	**−31.2**
	g CO_2-C kg^{-1} meat $year^{-1}$	**0.24**	**−0.96**	**−0.04**	**−0.1**	**−0.3**	**−0.14**	**−0.03**	**−0.40**

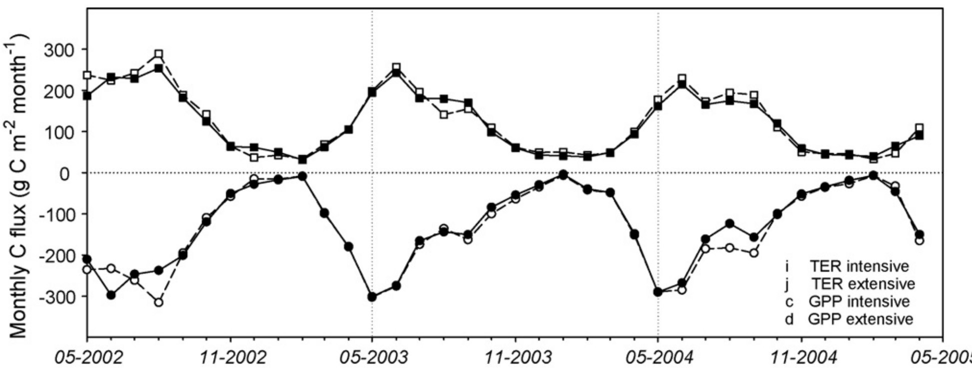

Fig. 4. Monthly sums of gross primary production (GPP, circles) and total ecosystem respiration (TER, squares) in the extensive (black) and intensive (white) paddocks.

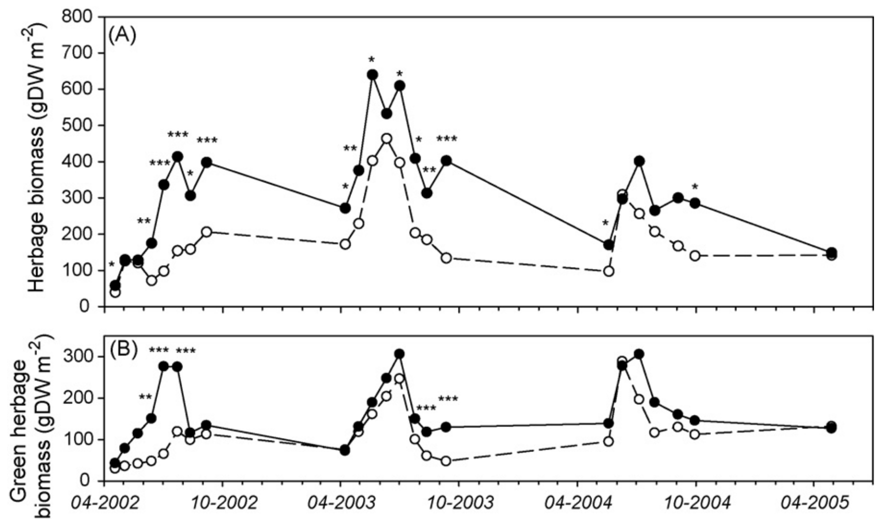

Fig. 5. Plant cover characteristics during the experimental period in the extensive (black) and intensive (white) paddocks. (A) Total herbage biomass and (B) green herbage biomass. *, **, ***Significant differences between the paddocks at the 0.5, 0.1 and 0.01 level, respectively.

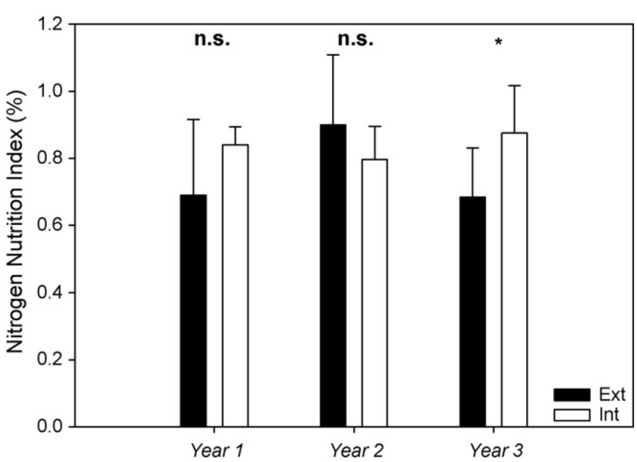

Fig. 6. Yearly average of nitrogen nutrition index in the extensive (full bars) and intensive (open bars) paddocks.

compared to the extensive paddock (9.8 and 5 g CH_4-C m^{-2} $year^{-1}$, respectively; Table 2).

3.6. Soil CH_4 and N_2O fluxes

The changes of CH_4 concentration within the static chambers were small and within the measurement error (data not shown). N_2O fluxes in the extensive paddock were consistently lower than in the intensive paddock (Fig. 8). For each paddock separately, the annual emissions were of the same order of magnitude in years 1 and 2, but were about half in year 3 (Table 1).

3.7. GHG budget

Annual GHG budgets were calculated in CO_2-C equivalents. Emissions of N_2O and CH_4 reduced by 89 and 55% the atmospheric sink activity of the intensive and extensive treatments, respectively. The GHG balance averaged across the study years was −10 and −31 g CO_2-C equivalents in the intensive and extensive paddock,

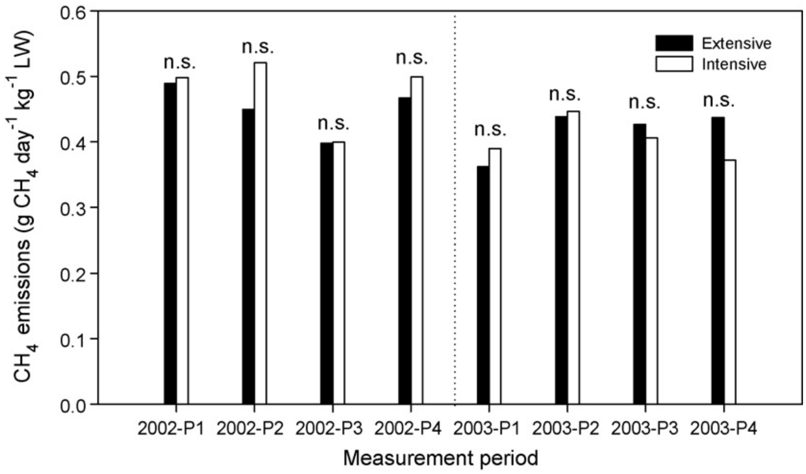

Fig. 7. Methane emissions by grazing cattle measured during eight campaigns in 2002 and 2003 in the extensive (full bars) and intensive (open bars) paddocks.

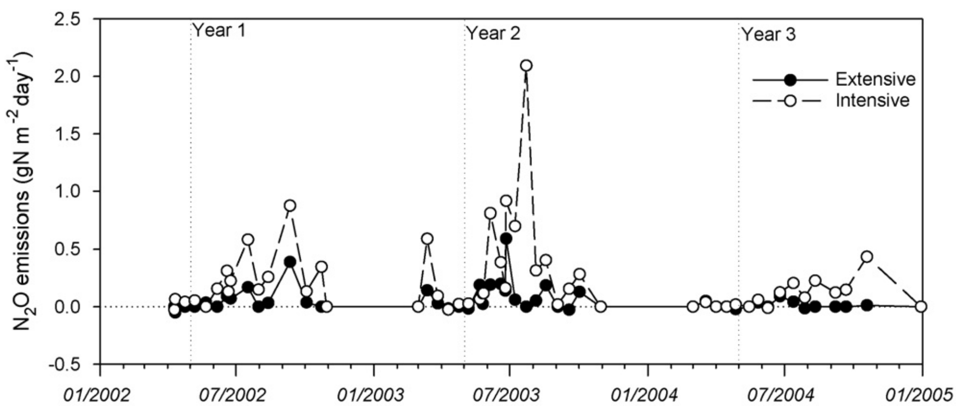

Fig. 8. Nitrous oxide emissions measured in accumulations chambers in the intensive (open circles) and extensive (full circles) paddocks. Lines figure the linear interpolation used for yearly N_2O budget calculation.

respectively (Table 2). In year 1, the intensive paddock showed a net GHG source activity of 37 g CO_2-C m^{-2} $year^{-1}$ while the extensive paddock displayed a strong sink activity (−67 g CO_2-C equivalents m^{-2} $year^{-1}$, Table 2). In year 2, both paddocks were small sinks for GHG. During the last experimental year, per unit ground area, the intensive paddock displayed a GHG sink three times greater than that of the extensive paddock.

The GHG budget was also calculated per unit of animal product, which is per kg of LW gained during the growing season. On average of the 3 years, the LW gain per hectare was twice as much in the intensive compared to the extensive paddock. Nevertheless, the GHG sink activity per unit LW gain was greater in the extensive compared to the intensive paddock (Table 2).

4. Discussion

In temperate grasslands, management choices to reduce GHG emissions may involve important trade-offs: for example, preserving grasslands and adapting their management to improve C sequestration in the soil may increase N_2O and CH_4 emissions (Soussana et al., 2004). We have investigated the consequences of two contrasting management options for the C and GHG budgets: an intensive grazing management, with high fertilizer N application and a high stocking density, versus an extensive grazing management without N added and at half the stocking density of the control. A significant limitation of the eddy covariance technique is that it does not allow for the replication of treatments. Below, we assume that all differences between the extensively and the intensively managed paddocks resulted from the contrast in management, which was initiated at the start of the experiment by dividing in two a large and homogeneously managed grassland paddock.

4.1. C budget in continuously grazed grasslands

In grazed-only grassland, herbivores recycle most of the ingested organic matter and, in the absence of organic C imports (i.e. manures), the C balance is expected to be close to equilibrium (Soussana et al., 2004). Under our experimental conditions, the annual NEE accounted for a small fraction of the GPP (between 3 and 10%) (Table 2). This differs significantly from relatively young temperate forests which exhibit much higher NEE/GPP ratios (e.g. NEE/GPP of 28% in two temperate deciduous forests, Jarvis et al., 2004). The biological basis for this discrepancy is worth noting: forest ecosystems store C in aboveground biomass before reaching full maturity, while there is usually no increase over years in the mean annual aboveground biomass of managed permanent grassland. In grazed systems, the herbage digestible organic C is mostly respired and only a very small fraction of the GPP is used for animal growth (ca. 0.1%, Table 2) or milk production. The non-digestible fraction (ca. 20–40%) of the ingested C returns to the soil (mainly as faeces).

Despite large interannual variability, the average annual NEE of the two grassland paddocks indicated a small apparent C sink activity (75–99 g C m^{-2} year^{-1}, Table 2), in agreement with results from other sites reviewed by Novick et al. (2004) and by Soussana et al. (2007).

Nevertheless, even under grazing, the NBP is not equal to the NEE (see Eq. (4)). Our study points at the need to include in the C budget animal related C fluxes (F_{CH_4} and F_{LW}) which are not measured by eddy covariance. The LW gain (F_{LW}) reached 0.9–1.7 g C m^{-2} year^{-1} and F_{CH_4} was estimated at 5–10 g C m^{-2} year^{-1}. Taken together, these two fluxes account for ca. 10% of the NEE value (Table 2). As a result, our estimates of NBP indicate that the net C storage in the grassland paddocks was less than 3% of the total organic C content of the top soil (6 kg C m^{-2} in the first 10 cm). Such a small rate of change cannot be detected within a few years by measuring changes in soil C stocks especially when spatial heterogeneity of the soil C stocks is accounted for (Lal et al., 2001).

A missing flux (Janssens et al., 2003) in our study is the dissolved organic and inorganic C (DOC/DIC) leaching. Siemens and Janssens (2003) estimated that 11 ± 8 g C m^{-2} year^{-1} of total dissolved C may be lost *via* leaching. Given the soil pH (5.5) at our site, DIC leaching should be low. A DOC leaching rate of 11 g C m^{-2} year^{-1} would affect (−13%, on average) the NBP to a similar extent as the inclusion of F_{CH_4} and F_{LW}, without changing the conclusion of a C sink activity in the two treatments.

The persistent C sink activity of both paddocks occurs despite the high organic matter content of the soil linked to its volcanic origin. Andosols are soils derived from volcanic ash and are characterized by large accumulations of organic C with mean C contents reaching ca. 25 kg C m^{-2} in the first meter (Batjes, 1996). Such large accumulations of organic matter are commonly explained by the protection of SOM by allophane (Mizota and Van Reeuwijk, 1989). Management effects of soil C stock in Andosols should thus be considered in relation to the large passive organic matter pool in these soils (Parfitt et al., 1997). While organic soils (histosols) can generate large fluxes of CO_2 to the atmosphere following a disturbance event (Ogle et al., 2003), SOC is relatively more stable in Andosols (Krull et al., 2001). Conversely, despite high C stocks, Andosols do not have a high potential of soil C sequestration under permanent vegetation (Feller et al., 2001).

4.2. Grazing management effects on the C budget

In a continuously grazed grassland, the aboveground herbage mass varies as the difference between growth, on the one hand, and senescence and animal defoliation on the other. Each year, the magnitude of the GPP and TER fluxes was always greater in the intensive compared to the extensive paddocks, indicating a faster C turnover under the intensive compared to the extensive grazing treatment.

The first experimental year was a transition between the previous management, dominated by hay cutting, and the experimental grazing treatments. Before the start of the experiment, the paddock was grassland characterized by high herbage utilization through cutting and low aboveground biomass at season restart (Fig. 5A). During the first year, the transition to grazing led to the accumulation of aboveground biomass (by 80 and 140 g C m^{-2} in the intensive and extensive paddocks, respectively (Fig. 5A)). The supplementary C trapped in the aboveground biomass accumulation in the extensive paddock (+60 g C m^{-2}, compared to the intensive treatment) may account for the lower NEE value in this treatment (−62 g C m^{-2} year^{-1} compared to the intensive treatment, Table 2). Functionally, this C stock variation can be considered as a transient effect of the differentiation between the two grazing treatments. In the two following years, both paddocks were close to steady state, at least with respect to aboveground biomass (Fig. 5A). Year 2 was characterized by a larger C sink in the intensive compared to the extensive treatment (Table 2). This additional sink occurred as the result of a higher annual GPP in the intensive compared to the extensive grazing treatment, while the difference in TER was smaller (Table 2). In year 3, the same trend was found with a supplement of 150 and 60 g C m^{-2} year^{-1} of GPP and TER, respectively, in the intensive compared to the extensive treatment (Table 2).

This result conflicts with the assumption by Parsons et al. (1983) of a declining GPP with increasing stocking density as a result of reduced light interception and leaf area index. There are two possible causes for this discrepancy: (i) fertilizer N supply in the intensive treatment may have stimulated GPP, as indicated by the higher nitrogen nutrition index of the herbage in the intensive compared to the extensive paddock in year 3 (Fig. 6) and (ii) there were more dead shoots in the extensive grazing treatments during summer (Fig. 5B) as well as a higher proportion of stems (data not shown) which were presumably less active in terms of photosynthesis than leaves.

Indeed, on a monthly basis, the between paddocks difference in GPP peaked during late summer (Fig. 4), when the extensively grazed canopy comprised a large proportion of stems and spikes. From these results, we believe that the extensive grazing management is likely to store less C in the longer term than the intensive treatment due to the conjunction of a lower NNI and of a lower proportion of photosynthetically active organs. Grassland management methods that abruptly reduce herbage use and fertilizer supply may therefore not be appropriate as a mitigation option.

4.3. Grassland GHG budget

CH_4 emissions per animal in the intensive and extensive paddock did not differ significantly at any of the eight measurement dates (Fig. 7). A detailed discussion about the effects of the treatments on the diet quality and herbage intake can be found in this issue (Pinarès-Patino et al., 2007). When CH_4 emissions are expressed in CO_2 equivalents, it appears that they strongly affected the GHG budget, on average by 82 and 42 g CO_2-C m^{-2} $year^{-1}$ in the intensive and extensive treatments, respectively (Table 2). Therefore, methane emissions induced a 56 and 82% offset of the NEE sink activity in the extensive and intensive grazing treatments, respectively. Keppler et al. (2006) have recently reported CH_4 emissions by plant tissues under aerobic conditions. With accumulation chambers placed on the soil and vegetation, we were not able to detect any significant change in the CH_4 concentration, compared to background level, during the 3 h of incubation in the dark. This result tends to show that in grazed grasslands, CH_4 emission by plant tissues is unlikely to affect significantly the ecosystem GHG budget.

N_2O emissions in grassland ecosystems are governed by numerous environmental and management drivers including soil type (Allen et al., 1996), soil water content and temperature (Clayton et al., 1997; Flechard et al., 2007), as well as the amount and type of N fertilizer supply (Clayton et al., 1997). Under our conditions, in the absence (extensive treatment) of N fertilizer supply, N_2O emissions were lower (0.04 g N m^{-2} $year^{-1}$, on average) than the ‘agricultural background’ value of 0.1 g N m^{-2} $year^{-1}$ reported by IPCC (2001). This result demonstrates the potential for low N_2O emissions from an extensively grazed grassland which does not receive N fertilizer (Fig. 8). Moreover, the emissions factors (see Flechard et al., 2007) were smaller (0.6, 0.4 and 0.2% in years 1, 2 and 3, respectively) than the default value (1.25%) reported by IPCC (2001). The same result has been obtained in other managed grassland sites (see Flechard et al., 2007). Some characteristics of Andosols may also contribute to the low N_2O emissions observed here. The extremely low bulk density of the soil in this experiment (640 kg m^{-3}) and its high total porosity (0.76 m^3 m^{-3}) compared to other European grassland sites (Flechard et al., 2007) induces water-filled pore space level that rarely exceeds 50–60% during the growing season. Nitrification is the main process contributing to N_2O emissions in Andosols (Granli and Bockman, 1994), as anaerobic conditions are less frequent in these soils (Akiyama and Tsuruta, 2002). Hence, N_2O emissions depend mostly on the ammonium-N source rather than on the nitrate-N source and may therefore be limited by low NH_4 concentrations (data not shown). On average over the 3 years, both paddocks acted as small GHG sinks with −10 and −31 g CO_2-C equivalents m^{-2} $year^{-1}$ in the intensive and extensive paddock, respectively (Table 2). Non-CO_2 GHG emissions induced a strong offset of the sink activity calculated based on NEE (−89% and −55% for the intensive and extensive paddock, respectively).

5. Conclusion

These results together with other studies of grassland C budget (Soussana et al., 2007; Gilmanov et al., 2007) show that European temperate grasslands act as net sink for atmospheric CO_2. In addition, our results clearly demonstrate the need for taking into account the fluxes of all three gases (CO_2, N_2O and CH_4) when calculating the greenhouse gas balance of a grassland. As expected, reducing fertilizer input and grazing pressure strongly reduced both CH_4 and N_2O emissions per unit land area. Nevertheless this management option seems to gradually reduce the C storage potential of the grassland. These findings support the hypothesis of a trade-off between mitigation of CH_4 and N_2O emissions and maintenance of C sink activity (Soussana et al., 2004). Hence, grassland management methods that abruptly reduce herbage use and fertilizer supply may not be appropriate as a mitigation option. Comparisons with other sites have been undertaken in order to assess whether the differences between treatments obtained in the present experiment can be generalized (Soussana et al., 2007; Gilmanov et al., 2007).

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