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# The greenhouse gas balance of European grasslands

**P. Ciais<sup>1</sup>, J. F. Soussana<sup>2</sup>, N. Vuichard<sup>1,3</sup>, S. Luysaert<sup>1,4</sup>, A. Don<sup>4</sup>,  
I. A. Janssens<sup>4</sup>, S. L. Piao<sup>6,1</sup>, R. Dechow<sup>5</sup>, J. Lathière<sup>1</sup>, F. Maignan<sup>1</sup>,  
M. Wattenbach<sup>7</sup>, P. Smith<sup>7</sup>, C. Ammann<sup>8</sup>, A. Freibauer<sup>5</sup>, E. D. Schulze<sup>5</sup>, and the  
CARBOEUROPE Synthesis Team\***

<sup>1</sup>Laboratoire des Sciences du Climat et de l'Environnement, CEA CNRS UVSQ, UMR 1572,  
91191 Gif sur Yvette, France

<sup>2</sup>Grassland Ecosystem Research, INRA, 234, Av. du Brézet, Clermont-Ferrand, 63039,  
France

<sup>3</sup>Centre International de Recherche sur l'Environnement et le Développement,  
Nogent-sur-Marne, France

<sup>4</sup>Department of Biology, University of Antwerpen, Universiteitsplein 1, 2610 Wilrijk, Belgium

<sup>5</sup>Max Planck Institute for Biogeochemistry, Hans-Knoell-Strasse 10, 07745 Jena, Germany

<sup>6</sup>Department of Ecology, Peking University, Beijing 100871, China

<sup>7</sup>Institute of Biological and Environmental Sciences, School of Biological Sciences, University  
of Aberdeen, Cruickshank Building, St. Machar Drive, Aberdeen, AB24 3UU, UK

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<sup>8</sup> Fed. Research Station ART, Reckenholzstr. 191, 8046 Zürich, Switzerland

\* G. Abril, O. Bouriaud, C. Beer, G. Churkina, J. Grace, M. Jung, I. Levin, G.-J. Nabuurs, J.-D. Paris, D. Papale, P. Peylin, M. Reichstein, M.-J. Schellhaas, M. Vetter, G.-R. van der Werf, K. Van Oost, N. Viovy, S. Zaehle

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Correspondence to: P. Ciais (philippe.ciais@cea.fr)

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## Abstract

The long-term carbon balance (NBP) of grasslands is estimated by combining scarce multi-year eddy-covariance observations at ecosystem observation sites where information on carbon inputs and harvesting removals is available. Following accounting for carbon leached to rivers, we estimated grasslands to be net carbon sinks of  $74 \pm 10 \text{ g C m}^{-2} \text{ yr}^{-1}$ . Uncertainties arise from the small number of sites and the short measurement period. Only 11 sites, out of a total of 20 grassland sites in Europe where eddy covariance systems are installed, were set-up for estimating NBP. These 11 selected sites are representative of intensive management practice and we lack information on disturbance history, such as plowing. This suggests that the grassland NBP estimate is likely biased towards overestimating the sink, compared to the European average. Direct measurements of Net Primary Productivity (NPP) are not possible in grasslands given permanent biomass removal by grazing and mowing, uncertainties in rhizodeposition and production of volatile organic carbon compounds lost to the atmosphere. Therefore, the grassland process-based ecosystem model PASIM was used to estimate the spatial-temporal distribution of NPP, providing a European average value of  $750 \pm 150 \text{ g C}$  across extensively grazed, intensively grazed pastures, and forage production systems. In Europe the NPP of grasslands seems higher than that of croplands and forests. The carbon sequestration efficiency of grasslands, defined as the ratio of NBP to NPP, amounts to  $0.09 \pm 0.10$ . Therefore, per unit of carbon input, grasslands sequester 3–4 times more carbon in the soil than forests do, making them a good candidate for managing onsite carbon sinks. When using the 100 yr greenhouse warming potential for  $\text{CH}_4$  and  $\text{N}_2\text{O}$ , their emissions due to management of grasslands together offset roughly 70–80% of the carbon sink. Uncertainties on the European grassland greenhouse gas balance, including  $\text{CO}_2$ ,  $\text{CH}_4$  and  $\text{N}_2\text{O}$  fluxes are likely to be reduced in the near future, with data being collected from more sites, and improved up-scaling methods.

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## 1 Introduction

Grassland soils are particularly rich in soil organic carbon (SOC), partly due to active rhizodeposition and to the activity of earthworms and arbuscular mycorrhizal fungi that promote macro-aggregate formation and consequently stabilize SOC for extended periods (Six et al., 2002; Bossuyt et al., 2005; Soussana et al., 2004; Wilson et al., 2009). However, these macro-aggregates can be destroyed by plowing, drought and high rainfall, and grassland SOC stocks may be vulnerable to changes in management regimes as well as to climate warming. Grasslands are not a natural type of ecosystem in Europe. They are continuously managed as multi-purpose systems for forage production (hereafter called meadows), animal grazing (hereafter called pastures) or a combination of both.

In a former assessment of the European carbon balance, Janssens et al. (2003) concluded that grasslands were the most uncertain component of the European-wide carbon balance in comparison to forests and croplands. They estimated a net grassland carbon sink of  $66 \pm 90 \text{ g C m}^{-2} \text{ yr}^{-1}$  over Geographic Europe, for a grassland area of  $151 \times 10^6 \text{ km}^2$ . However, this estimate was not based on actual measurements, but on a simple model using yield census and land use information. This model was developed for modeling climate change impacts, and not to determine the baseline carbon flux related to management and climate (Vleeshouwers and Verhagen, 2002). Other regional estimates of grassland carbon balance were made, based upon soil C inventories (Bellamy et al., 2005), but these studies do not provide European coverage.

Several new grassland eddy-covariance observation sites were initiated in Europe, with support of EU and national projects, such as GREENGRASS (Soussana et al., 2007; <http://www2.clermont.inra.fr/greengrass/>), CARBOMONT (<http://www.uibk.ac.at/carbomont/>; Bahn et al., 2008; Rogiers et al., 2005, 2008; Wohlfahrt et al., 2005, 2008a,b; Soussana et al., 2007) and CARBOEUROPE-IP (<http://www.carboeurope.org/>; Barcza et al., 2003). These sites provide new data for constraining the carbon balance of grasslands, and for supporting the development and calibration

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of more realistic models. These sites cover different types and intensities of management, opening a new window to understand grassland carbon cycling and trade-offs between CO<sub>2</sub> sinks and CH<sub>4</sub> and N<sub>2</sub>O emissions. However, eddy covariance methods only measure the net flux of CO<sub>2</sub> and do not account for lateral input and export of carbon. Therefore, work must be done to relate ecosystem CO<sub>2</sub> flux measurements to the carbon balance, or Net Biome Productivity (NBP).

Models accounting for pasture management have been developed recently (Smith et al., 2005; Bondeau et al., 2007; Vuichard et al., 2007a). To date, however, no such model is able to integrate realistically the effects of land use and management drivers, combined with climate and CO<sub>2</sub> changes on carbon cycling. Therefore, all component fluxes of the grassland carbon balance cannot be assessed within a single coherent modeling framework. The goal of this study is to provide a synthesis of the carbon fluxes, carbon balance, and greenhouse gas balance of grasslands in the EU-25. To meet this goal, we selected data from the few eddy-covariance sites where multi-year records are available and the lateral import and export of carbon measured in addition to eddy-covariance net CO<sub>2</sub> flux (NEE). We selected 11 such “golden sites” where the carbon balance (NBP) can be constrained, out of 20 eddy-covariance sites in operation in Europe. These data are combined/up-scaled using GPP from a process-based model across EU-25, in order to seek answers to the following questions:

- Do recent eddy-covariance observations and site-level data support the former estimate of a carbon sink in European grasslands – and what are the uncertainties?
- How does the Net Primary Productivity (NPP) of grasslands vary between grazing and cutting regimes across Europe?
- What fraction of NPP is stored in soils in European grasslands?
- To what degree do the emissions of methane (CH<sub>4</sub>) and nitrous oxide (N<sub>2</sub>O) offset carbon sequestration in grassland soils?

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In Sect. 2, the methods and data used to derive new estimates of the carbon balance and net primary productivity are detailed, as well as calculations of CH<sub>4</sub> and N<sub>2</sub>O emissions. The results for each flux are presented in Sect. 3. Uncertainties and comparison with previous estimates are given in Sect. 4; conclusions are drawn in Sect. 5.

## 2 Material and methods

### 2.1 Study area

In this study, our definition of grassland follows the Land Use/Cover Area Frame Statistical Survey (LUCAS) (European Commission, 2003; Kempen et al., 2007). Europe is defined as the 25 member states of the European Union (EU-25; d.d. 31 December 2006) i.e. Austria, Belgium, Cyprus, Czech Republic, Denmark, Estonia, Finland, France, Germany, Greece, Hungary, Ireland, Italy, Latvia, Lithuania, Luxembourg, Malta, Netherlands, Poland, Portugal, Slovakia, Slovenia, Spain, Sweden and UK. Grasslands cover  $0.57 \times 10^6 \text{ km}^2$  that is 15% of the EU-25 territory, and 35% of the EU-25 agricultural lands.

The studied grasslands are pastures (grazed) or meadows (cut) or in most cases a mixture of both types. Shrublands and temporary grasslands established in-between crop rotations are not included in this study. The area of grassland adopted here is lower than in other studies. For instance the CORINE/PELCOM satellite-based land cover dataset gives a grassland area 15% higher than our estimate ( $0.83 \times 10^6 \text{ km}^2$ ). The Food and Agriculture Organization (FAO) estimates an area 100% higher ( $1.5 \times 10^6 \text{ km}^2$ ) for total Eastern and Western Europe FAO (2006). The choice of a particular definition of grasslands has no impact on our flux estimates, because the methods that we use estimates primarily fluxes per unit area, except for N<sub>2</sub>O and CH<sub>4</sub> emissions estimated from national-scale data (Sect. 2.4). Most numbers reported in this manuscript represent mean values for the last decade, except otherwise specified. Wherever needed to better understand regional details in the grassland carbon cycle, regionally different grassland ecosystems are discussed separately.

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## 2.2 Components of grasslands carbon balance

We refer here to the definition of an ecosystem long-term carbon balance (Schulze and Heimann, 1998), or Net Biome Productivity given in Appendix A (Eq. A5).

$$\text{NBP} = -\text{NEE} + I - H - F_{\text{Animal Products}} - F_{\text{CH}_4-\text{C}} - F_{\text{VOC}-\text{C}} - F_{\text{fires}} - E - D \quad (1)$$

Where NEE is the net ecosystem exchange CO<sub>2</sub> flux measured by eddy-covariance systems. The convention is that NEE is positive if an ecosystem loses CO<sub>2</sub>. The flux  $I$  denotes the lateral carbon input by manure and slurry addition, and  $H$  the flux of biomass cut and exported from the ecosystem.  $F_{\text{CH}_4-\text{C}}$  is the total methane flux in C units, lost to atmosphere through grazing animal enteric fermentation.  $F_{\text{VOC}-\text{C}}$  is the emission of Volatile Organic Compounds (VOC) in C units, excluding CH<sub>4</sub>.  $F$  corresponds to fire emissions to the atmosphere.  $E$  is the lateral removal/accumulation of soil carbon from erosion/re-deposition processes.  $D$  is the net flux of dissolved organic carbon (DOC) leached to rivers by grassland soils.

The system boundary for NBP is the grassland ecosystem, so that all the carbon in the harvested flux  $H$  is considered here lost by the system, even though this carbon will be returned to the atmosphere at the farm or regional scale, where the cut grass biomass is used. Because the definition of NBP in Eq. (1) does not include the fate of exported C,  $F_{\text{CH}_4-\text{C}}$  does not include methane released by cattle in farm buildings using carbon from  $H$ . By contrast, CO<sub>2</sub> respired by grazing animals is assumed to be measured by eddy-covariance NEE. The enteric fermentation at grazing is included in  $F_{\text{CH}_4-\text{C}}$ . Grazing does not export C from the system, apart from animal products like meat and milk, in the flux  $F_{\text{Animal Products}}$ , which are small C fluxes. Carbon exports in animal products reach only between 2 and 20% of C intake for meat and milk production, respectively. The flux  $F_{\text{Animal Products}}$  will thus be neglected in the following. The specificities of grasslands with respect to Eq. (1) are:

- Grassland is the only ecosystem associated to significant CH<sub>4</sub> emissions from grazing animals. Even though CH<sub>4</sub> is a potent greenhouse gas and its emissions

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intervene in the net greenhouse gas balance, the flux  $F_{\text{CH}_4-\text{C}}$  is very small on a carbon mass basis, and hence will be neglected for calculating NBP in Eq. (1).

- Fire emissions ( $F_{\text{fires}}$ ) from grasslands are also very small in Europe. We obtain from the global GFEDV2.0 fire emission dataset based on satellite burned areas observation  $F=0.3 \text{ g C m}^{-2} \text{ yr}^{-1}$  over 1997–2006 (Van der Werf et al., 2006).  $F$  is only 0.1% of the flux lost by harvesting forage grass. In the EU-25, grasslands fires emission is one order of magnitude smaller than cropland (Ciais et al., 2010) and forest fires emissions (Luysaert et al., 2010), and  $F$  is thus neglected in Eq. (1).
- The erosion carbon flux ( $E$ ) is the sum of human accelerated erosion processes and natural processes such as landslides or heavy rainfalls. Unlike in cropland soils, erosion is negligible in grasslands, except in Southern European pastures (see Van Oost et al., 2007; their Tables S2 and S3), hence  $E$  is also neglected in Eq. (1).
- Part of the assimilates is converted to a wide spectrum of VOC that are emitted to the atmosphere ( $F_{\text{VOC}-\text{C}}$ ). Apart from few pilot studies (Brunner et al., 2007; Davison et al., 2008) long-term flux measurements (e.g. by proton transfer mass reaction) of a nearly complete range of VOC are to date insufficient to produce a measurement-based European-wide estimate of VOC emissions. In this study, the results from the global model of VOC emission of Lathière et al. (2006) are used for quantifying the isoprene and monoterpene flux (see below for details). But in presence of large uncertainties,  $F_{\text{VOC}-\text{C}}$  will be neglected in Eq. (1).
- Dissolved organic carbon (DOC) constitutes only a minor fraction of the vast carbon stores in grassland soils. During periods with positive water balance, part of this DOC is leached out of the soil profile and eventually transferred to rivers. In contrast to forests and surface waters, studies on DOC exports from grasslands are rare (McTiernan et al., 2001; Don and Schulze, 2008). Nonetheless, the few



available studies have indicated non-negligible DOC losses, and we therefore included this term in our calculations of NBP (see below for details).

## 2.3 Carbon fluxes

Most of the results presented in this study are based on the data of Table 2 from published literature and model results (Fig. 1). References to these data or models are given in Table 1. When other dataset were gathered to complete analyses or discussion, in-text citations are given. Because we bring together results from a variety of methods, there are inevitable inconsistencies in datasets and methodologies.

### 2.3.1 NBP from eddy-covariance sites

About 20 grassland sites equipped with eddy covariance instrumentation are providing online measurements of NEE across Europe (Gilmanov et al., 2007). The observed flux  $NEE_{obs}$  is separated into  $GPP_{obs}$  and  $Re_{obs}$  using assumptions on night-time respiration controls (Reichstein et al., 2005). The climatic drivers of NEE, GPP and Re were analyzed at these 20 sites by Gilmanov et al. (2007). For managed grasslands however, NBP does not equal  $NEE_{obs}$ , and additionally includes lateral carbon inputs and exports (Soussana et al., 2007) as shown by Eq. (1). For instance, harvested biomass is not included in  $NEE_{obs}$  and thus induces a difference between NBP and  $NEE_{obs}$ . In this study, we compiled data on carbon input and export fluxes at 14 sites out of the 20 used by Gilmanov et al., in order to infer NBP from NEE. The sites where NBP can be estimated are listed in Table 2. We further excluded from the analysis two Irish sites (Table 2) where the amount of manure/slurry input was too uncertain, and a Danish site (Table 2) where a massive amount of manure was applied during the observation period. This leaves us with 11 “golden sites” covering part of the period 2002–2006. At each site,  $NBP_{obs}$  is determined from the mean value of eddy-covariance NEE during the observation period, corrected for import and export C fluxes. Leaching of carbon to rivers was not measured at each site, and we assumed a default flux of carbon

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exported to rivers of  $10 \text{ g C m}^{-2} \text{ yr}^{-1}$  which includes DOC, DIC and POC, even though DOC is expected to be a dominant and yet elusive component of this flux. This flux is derived from C-concentrations in small watershed with known lithology and soil carbon concentration, extrapolated at the European scale (Ciais et al., 2008). An uncertainty of 50% is assumed for this flux. At site level, measurements of DOC fluxes are highly variable across sites, depending upon soil depth and clay content, giving a range 8 to  $50 \text{ g C m}^{-2} \text{ yr}^{-1}$  (Don and Schulze, 2008).

The average value of  $\text{NBP}_{\text{obs}}$  across the 11 European grassland “golden sites” is given by:

$$\text{NBP}_{\text{obs}} = \frac{1}{N} \sum_{i=1}^N \left[ \overline{\text{NEE}}_{\text{obs}}(i) + I(i) - H(i) \right] - D \quad (2)$$

Where  $N$  is the number of sites and  $\overline{\text{NEE}}_{\text{obs}}(i)$  the temporal mean of 1/2-h eddy covariance measurements over the observation period at the  $i$ th site. Uncertainty is calculated by error propagation in Eqs. (1) and (2).

### 2.3.2 NBP from eddy-covariance sites up-scaled using process-based model (PASIM)

The PASTure Simulation Model (PASIM) is process-model of carbon and nitrogen cycling in grassland that was applied at the scale of Europe (Riedo et al., 1998; Vuichard et al., 2007a,b). PASIM results were evaluated against three of eddy-covariance grassland sites from Table 2 (Vuichard et al., 2007a), and against satellite vegetation greenness index (Vuichard et al., 2007b). The model reproduced the general features of the validation data and was hence considered suitable for up-scaling from site to region and Europe.

A simulation of NBP could not be done with PASIM in the set-up established by Vuichard et al. (2007b) and used here, because the model was run during

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a 10 000 yr spin up until fluxes and C pools reached steady state equilibrium, defining NBP=0. We used the GPP geospatial distribution from PASIM, differentiated between pastures ( $f_{\text{pasture}}$ ) and meadows ( $f_{\text{meadow}}$ ), in order to extrapolate the point-scale NBP<sub>obs</sub> measurements from flux towers. The mean EU-25 simulated GPP<sub>PASIM</sub> is of 1343±268 g C m<sup>-2</sup> yr<sup>-1</sup>. We assume the same relative error (20%) for GPP<sub>PASIM</sub> than for NPP<sub>PASIM</sub> (see details below). The observed site-level ratios (NBP/GPP)<sub>obs</sub> at each site are then combined with GPP<sub>PASIM</sub> according to:

$$\begin{aligned} \text{NBP}_{\text{obs}}^* = & \frac{1}{N_{\text{pasture}}} \sum_{i=1}^{N_{\text{pasture}}} \left( \frac{\text{NBP}}{\text{GPP}}(i) \right)_{\text{obs pasture}} \times f_{\text{pasture}} \times \text{GPP}_{\text{PASIM pasture}} \\ & + \frac{1}{N_{\text{meadow}}} \sum_{i=1}^{N_{\text{meadow}}} \left( \frac{\text{NBP}}{\text{GPP}}(i) \right)_{\text{obs meadow}} \times f_{\text{meadow}} \times \text{GPP}_{\text{PASIM meadow}} \end{aligned} \quad (3)$$

The rationale for justifying Eq. (3) as an alternative to Eq. (2) is that ratios are more robust than local NBP measurements for up-scaling from flux-tower sites to the continent. Values of the ratio (NBP/GPP)<sub>obs</sub> are observed to range from 0.01 to 0.24 at pasture sites, and from 0.02 to 0.05 at meadow sites (Soussana et al., 2007; Gilmanov et al., 2007). Removing one year randomly at each site to calculate NBP, defines an uncertainty on NBP from Eq. (3) on the order of 100%.

### 2.3.3 NBP contributed by historical land use change

Little is known about land use change impacts on grassland NBP at continental scale. Our sample of 11 sites contains grasslands of different ages, and so NBP<sub>obs</sub> does partially reflect some age factors. However, no flux-measurement chronosequence for grasslands after disturbance (e.g. after plowing and sowing of a new grassland) is available to analyze the relationship between NEE and age, and multiply this curve by changing grassland areas (see for forest Magnani et al., 2007). In absence of direct observations, we rely on a long term simulation of the ORCHIDEE vegetation model (Krinner et al., 2005) prescribed with changing grasslands area, to estimate the land-use change impacts on NBP. ORCHIDEE was first integrated during 10 000 yr until

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carbon pools in each grid point reached their steady state equilibrium value. This spin-up was performed by recycling climate fields reconstructed for the period 1901–1910 (Mitchell and Jones, 2005) and fixing atmospheric CO<sub>2</sub> and land cover to their value in 1860 (Ramankutty and Foley, 1999; Piao et al., 2009). Based on this initial state, two simulations (S1 and S2) are carried out from 1860 to 2002, to calculate the net carbon balance of each grid point. In simulation S1, only CO<sub>2</sub> and climate vary. In simulation S2, climate, CO<sub>2</sub> and land use vary (prescribed annual land cover use maps are accessible at <http://www.cnrm.meteo.fr/ensembles/>). The effect of historical land use change on NBP is assessed from the NBP difference between S2 and S1. The land use change effects on grassland NBP is defined by the space-time integral of modeled NBP over all grid points of EU-25 which saw either *newly established* grasslands or *abandoned grasslands* during the last 20 yr. This integral is divided by the total grassland area in EU-25 and by 20 yr, to obtain a NBP flux density.

### 2.3.4 NPP from eddy covariance data combined with process-based model (PASIM)

Insufficient NPP estimates were available for grasslands to produce a pure data-driven European-scale NPP estimate. An alternative data-oriented approach was applied, in which we combined the eddy-covariance observations of GPP with maps of the ratio of NPP/GPP (carbon use efficiency) obtained with the PASIM model applied over EU-25 (Vuichard et al., 2007b). This gives a data oriented estimate of NPP, called NPP<sub>obs</sub><sup>\*</sup> and defined by:

$$NPP_{obs}^* = \frac{1}{N} \sum_{i=1}^N \left( \frac{NPP_{PASIM}(i)}{GPP_{PASIM}(i)} \times GPP_{obs}(i) \right). \quad (4)$$

### 2.3.5 NPP from process-based model (PASIM)

Net primary productivity was modeled across the EU-25 on a grid of 1° by 1° using PASIM. In each grid point, NPP<sub>PASIM</sub> is simulated separately for cut and grazed grass-

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lands, and “mixed” between these two ecosystems using a simple management algorithm assuming the highest sustainable animal density (more details in Vuichard et al., 2007b). In order to account for N-fertilizer additions on NPP, two PASIM simulations are performed: a run called FER where grasslands receive an amount of N-fertilizer equal to the national average (FAO, 2002), and a run called NOFER where grasslands are not fertilized. NPP from these two simulations is combined into a single map, using ratios of fertilized to non-fertilized grasslands (FAO, 2002).

### 2.3.6 NPP from PASIM del corrected for extensive management

We expect  $NPP_{PASIM}$  to overestimate NPP in general, because the simulation used maximizes animal density in each grid point, which corresponds to a highly intensive management scenario everywhere. To better account for extensive management practices, we used the EUROSTAT NewCronos database (<http://epp.eurostat.ec.europa.eu>) where grassland is classified into *pasture*, *meadows excluding rough grazing* (i.e. intensively managed meadows) or *rough grazing* (i.e. extensively managed meadows). This information on management intensity is available in NewCronos at the scale of NUTS-2 regions, which are relatively coarse administrative zoning units (115 NUTS-2 regions in the EU-15). Moreover, these data originate from a sample survey only (Farm Structure Survey FSS-1997) and not from a census. This regional-scale intensive vs. extensive management information give by NewCronos over EU-15, is completed by national-scale information from the FAOSTAT database over the EU-25 (FAO, 2002). Overall, we found that 66% of the grasslands are *intensively* managed and 34% *extensively* managed. The ratio of intensively to extensively managed areas in each country is shown in Fig. 2. Using this information, NPP from PASIM was corrected in each grid point to account for extensive management, according to:

$$NPP_{PASIM}^* = y(X) \times NPP_{PASIM} + (1 - y(X)) \times NPP_{PASIM \text{ meadows NOFER}} \quad (5)$$

Where  $NPP_{PASIM \text{ meadows NOFER}}$  corresponds to the model run without fertilizer application chosen to approximate NPP of *extensively* managed grassland,  $y(X)$  is the fraction

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of intensively managed grasslands in each grid point from the NewCronos database, and  $X$  is the space coordinate.

### 2.3.7 NPP evaluation against data and uncertainty

There is no observational dataset that can be used to evaluate the above PASIM estimates of NPP. Nonetheless, for separate components of NPP, PASIM results were compared with field observations. The simulated harvest  $H$  of meadows (cut), which is the main component of above-ground NPP, was checked against site-level yield data from 12 sites of the FAO-Lowland Grassland Network (FAO, 2002). The fraction of NPP consumed by herbivores in pastures (yield) was indirectly evaluated against national statistics of animal loads by Vuichard et al. (2007b). In addition, leaf area index simulated was compared to measurements at three sites (Table 2) (Vuichard et al., 2007a) giving an average model error of 20% for yield and biomass. This relative uncertainty is used in the following to define the error of  $NPP_{PASIM}$  (Fig. 3).

### 2.3.8 $R_h$ from mass balance calculation

Heterotrophic respiration ( $R_h$ ) is extremely difficult to measure directly at the ecosystem level, just like NPP. Too few  $R_h$  data are available to estimate a European mean value. Using eddy-covariance measurements, NEE can be decomposed into GPP and  $R_e$  (Gilmanov et al., 2007; Reichstein et al., 2005). But NBP cannot be decomposed into NPP and  $R_h$ . In this study, we estimated  $R_h$  from mass balance considerations, according to:

$$R_h = NPP_{PASIM} - NBP_{obs} + I - H_{PASIM} - D \quad (6)$$

Where NPP and  $H$  are simulated by PASIM across EU-25 (see above) and  $NBP_{obs}$  is obtained from Eq. (2).

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## 2.4 N<sub>2</sub>O and CH<sub>4</sub> emissions estimates

Estimates of N<sub>2</sub>O emissions from grassland soils were calculated by three methods: 1) national UNFCCC statistics, 2) a fuzzy logic model (Dechow, in preparation) and 3) the PASIM model. Only direct emissions of N<sub>2</sub>O are considered here. Accounting for indirect N<sub>2</sub>O emissions, which are also included in the IPCC (2006) guidelines, would increase emissions by 2–5% (Crutzen et al., 2008; Davidson et al., 2009).

### 2.4.1 N<sub>2</sub>O emissions from disaggregation of UNFCCC statistics

UNFCCC statistics report national N<sub>2</sub>O emissions from agricultural soils by different sectors, but do not distinguish between grassland and cropland emissions. N<sub>2</sub>O emission was split as follows into cropland and grassland soils, respectively. The categories *synthetic fertilizer related emissions* (4.D.1.1 in UNFCCC nomenclature), N<sub>2</sub>O emissions falling into the *animal manure* category (4.D.1.2) and *nitrogen fixation* emissions (4.D.1.3) were split according to rules for region- and crop type-specific nitrogen demand on cropland with the remainder being allocated to grassland. This procedure follows state-of-the-art approaches by Freibauer (2003) and CAPRI Dynaspat. The resulting allocation factors for N input to grassland vary between less than 20% in Hungary, Finland and Sweden to more than 50% of the N input in Austria, Germany, and the Netherlands. N<sub>2</sub>O emissions from *histosols* (4.D.1.5) were attributed to croplands according to the emissions from cropland and grassland by country in Drösler et al. (2008). On average, 57% of the total N<sub>2</sub>O emissions from agricultural *histosols* in EU-25 were assigned to grasslands, and the rest attributed to croplands. N<sub>2</sub>O emissions falling in the *other direct and indirect emissions* categories of the UNFCCC were all attributed to forest, the main unfertilized land use type that receives atmospheric nitrogen deposition. *Pasture emissions* from grazing animals (4.D.2) were all attributed to grasslands. UNFCCC statistics for the two reference years 1990 and 2000 have been used.

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## 2.4.2 N<sub>2</sub>O emissions from fuzzy logic model

The fuzzy logic model (*Dechow and Freibauer, unpublished data*) is a sequence of “IF-THEN” rules that aims to estimate N<sub>2</sub>O emissions based on a combination of input factors. Model training finds the most suitable combination of information about soil properties (texture, organic carbon, organic nitrogen), climate conditions and management options (cutting regime, amount of mean applied N, type of applied fertilizer) in order to match annual emissions known at site level. A set of calibration data coming from 24 sites with 88 variants was extracted from a database described in Stehfest and Bouwman (2006). Cross-validation was performed by excluding groups of the same site from the calibration dataset ( $R^2=0.58$ ).

Factors used in the up-scaling procedure were the amount of applied N weighted by the type of fertilizer (organic or inorganic), mean autumn precipitation, winter temperature and difference of summer and winter temperature. A linear relationship between annual amount of applied N and nitrous oxide emissions was assumed for which the climate conditions determined slope and offset. Nitrogen addition via fertilizer in 1990 to 1999 was extrapolated from CAPRI DynaSpat data for the year 2000 (Leip et al., 2008) and country budgets from the EUROSTAT database.

Seasonal precipitation and temperature are from simulations with the REMO model (Vetter et al., 2007). Local distribution of grassland areas for year 2000 originates from a two step regression approach taking environmental factors (e.g. climate, soil properties and land cover), statistical data of the CAPRI database with information at NUTS 2 Level and the Land Use/Cover Area Frame Statistical Survey (LUCAS) into consideration (Leip et al., 2008; Kempen et al., 2007). This data is extrapolated to the time period 1990–1999 using statistics from FAO. The estimate we provide is the mean value over the 1990’s decade.

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### 2.4.3 N<sub>2</sub>O emissions from PASIM model.

The PASIM model simulations described above include nitrogen cycling processes in grasslands and associated N<sub>2</sub>O emissions. In PASIM, the main driver of N<sub>2</sub>O emission are the timing and mean annual amount of applied N-fertilizer, as well as local soil and climate conditions. The amount of N-fertilizers received by each grid point was estimated from national statistics combined with a modeled ratio of *fertilized* to *non-fertilized* grasslands (see Sect. 2.3). We did not consider organic N-applications since relevant information was not available at the European level, which causes N<sub>2</sub>O emissions to be under-estimated. In the grazing simulations, dung and urine were accounted for and assumed to be recycled in the soil. The estimate we provide is the mean value for year 1993.

### 2.4.4 Livestock CH<sub>4</sub> emissions from UNFCCC

In this section, the system boundary is the livestock, hence CH<sub>4</sub> emissions from both grazing animals and from animals in farm building fed with harvested forage grass, unlike in the NBP definition of Eq. (1) where the ecosystem was used as the system boundary. Methane emissions from enteric fermentation by grazing animals are derived from UNFCCC statistics (<http://unfccc.int/di/DetailledByParty/Setup.do>), based on emission factors by animal categories. By year 2000, the reported EU-25 cattle population amounts to 65 millions Livestock Units (EUROSTAT, 2005). Note the relatively large uncertainty in this number when expressed into livestock units. According to UNFCCC, there are 27 million dairy cattle of 570 kg/head weight and 68 million with 377 kg/head. Therefore, assuming from these UNFCCC data that 500 kg corresponds to one average Livestock Unit, then this would give 83 million Livestock Unit in EU-25 instead of 65 millions in the EUROSTAT Census. This implies a 30% error on livestock CH<sub>4</sub> emissions estimated using this method.

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## 3 Results

### 3.1 Net primary productivity

#### 3.1.1 PASIM NPP

Figure 4 provides the spatial distribution of grassland NPP across Europe, from PASIM. The average NPP of pastures and meadows is 1237 and 646  $\text{g C m}^{-2} \text{yr}^{-1}$ , respectively. Figure 4a and b shows that the regional patterns of NPP are quite similar between pasture and meadow. NPP is higher around the North-western Atlantic coast and lower in southern and eastern Europe. The fact that pasture NPP is two times higher than meadows NPP in the model reflects more intense nitrogen recycling. If we take the fractional cover of pasture (27%) vs. meadows (73%) calculated by PASIM over Europe under the assumption that farmers everywhere maximize the number of animals per unit area (Vuichard et al., 2007b), then the combined all-grasslands NPP amounts to 797  $\text{g C m}^{-2} \text{yr}^{-1}$  with an uncertainty of 20% (Fig. 3). In addition, if we correct this modeled NPP for extensive management according to Eq. (5), we obtain a “best value” of 750  $\text{g C m}^{-2} \text{yr}^{-1}$  (Fig. 3). The mean NPP corrected for extensive management is rather similar (7% lower) to the standard NPP of PASIM, but regional differences increase when going down to specific regions, in particular over southern European regions where the corrected NPP is up to 30% lower than the standard NPP. For comparison, the ORCHIDEE process-based model without grassland management, provides over Europe  $\text{NPP}=685 \text{ g C m}^{-2} \text{yr}^{-1}$ .

#### 3.1.2 Data oriented NPP

A data-oriented NPP estimate, here called  $\text{NPP}_{\text{obs}}^*$  is obtained by multiplying the observed eddy-covariance  $\text{GPP}_{\text{obs}}$  by the modeled ratio  $\text{NPP}/\text{GPP}_{\text{PASIM}}$  at each of the 11 “golden” sites (Table 2). Given an average  $\text{GPP}_{\text{obs}}$  of  $1279 \pm 160 \text{ g C m}^{-2} \text{yr}^{-1}$  and a ratio  $\text{NPP}/\text{GPP}_{\text{PASIM}}$  of 0.59; this translates into  $\text{NPP}_{\text{obs}}^* = 758 \text{ g C m}^{-2} \text{yr}^{-1}$ . If we take

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instead a ratio  $NPP/GPP_{obs}=0.46$ , inferred from the Klumpp et al. (2007) isotopic labeling experiment in pasture monoliths, we would obtain  $NPP^*_{obs}=588 \text{ g C m}^{-2} \text{ yr}^{-1}$ . This range of  $NPP^*_{obs}$  from 588 to  $758 \text{ g C m}^{-2} \text{ yr}^{-1}$  is representative of uncertainties in the ratio of NPP to GPP, i.e. the fraction of GPP respired by Ra in different ecosystems. For instance in a recent synthesis of NPP and GPP measurements at forest ecosystems, Piao et al. (2009) reported a range of 0.4 to 0.8 for the NPP to GPP ratio.

### 3.2 Carbon harvested and exported by mowing

Grassland harvest by cutting ( $H$ ) takes place during the growing season. We remind here that in this study grazing of pasture is not considered in this study to export C from the ecosystem, except for a negligible animal product flux (see details above). The  $H$  removals from cutting are more intimately linked to the seasonal dynamics of NPP for grasslands than for any other ecosystems. Using the PASIM model results, we obtain a ratio of  $H$  to NPP of 0.4. Across the EU-25, this gives  $H=263 \text{ g C m}^{-2} \text{ yr}^{-1}$  (Fig. 1). Using NPP from PASIM corrected for extensive management, a 7% reduction should be applied to this flux. For comparison, the grazed fraction of total NPP is only 0.18 in pasture, compared to a cut fraction of 0.4 in meadows. The  $H$  removals are therefore larger and more sporadic in cut meadows, which likely contributes to maintain their NPP lower. The spatial distribution of the  $H/NPP$  ratio across the continent is mainly controlled by climate conditions, in particular by the growing season length (more details in Vuichard et al., 2007b).

### 3.3 Heterotrophic respiration

Heterotrophic respiration  $R_h$  obtained from Eq. (6) amounts to  $486 \text{ g C m}^{-2} \text{ yr}^{-1}$ . Given eddy covariance  $GPP_{obs}$  and  $NEE_{obs}$  estimates,  $R_h$  is therefore equivalent to 40% of the total ecosystem respiration  $R_e$ . This average ratio  $R_h/R_e$  is lower than, but within the errors of the value determined experimentally by Byrne and Kiely (2006) at an intensively managed grassland in Ireland (56%). Overall,  $R_h$  in European grasslands

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represents 64% of NPP, given that 30% of NPP is exported away from the ecosystem via  $H$  and eventually respired outside the ecosystem. The remaining 6% is laid off to the soil as litter, available for NBP.

### 3.4 Net ecosystem exchange

5 Using eddy-covariance measurements from the 11 sites, we estimate  $NEE_{obs} = -240 \pm 70 \text{ g C m}^{-2} \text{ yr}^{-1}$  (Soussana et al., 2007). With 20 sites representing 26 site-years, Gilmanov et al. (2007) inferred  $NEE_{obs} = -150 \pm 200 \text{ g C m}^{-2} \text{ yr}^{-1}$ . Our higher value reflects the bias of the 11 golden sites towards more intensive management and long-growing seasons. This  $NEE_{obs}$  estimate and its associated uncertainty  
10 need to be refined, but confirms the existence of a permanent atmospheric  $\text{CO}_2$  sink in grasslands. By contrast, the ORCHIDEE model (Krinner et al., 2005) applied over Europe and without biomass export from grasslands calculates  $NEE = 25 \text{ g C m}^{-2} \text{ yr}^{-1}$  as the difference between NPP and RH, both fluxes being driven by  $\text{CO}_2$  and climate change. Such a  $\text{CO}_2$  sink corresponds by no means to C sequestration. A large  
15 fraction of  $\text{CO}_2$  removed from the atmosphere by NEE is accumulated in biomass, which gets harvested and exported, and released back as  $\text{CO}_2$  at farm scale or at regional scale.

### 3.5 VOC emissions

This term has not been derived from measurements yet, and was thus neglected in our calculation of NBP. For information however, the global process-based vegetation  
20 model of VOC emissions of Lathière et al. (2006) designed for atmospheric chemistry studies was sampled over EU-25 grasslands grid points. The inferred mean flux  $F_{VOC-C}$  is a source of  $50 \text{ g C m}^{-2} \text{ yr}^{-1}$  to the atmosphere, with  $10 \text{ g C m}^{-2}$  of monoterpene emissions and  $40 \text{ g C m}^{-2} \text{ yr}^{-1}$  of isoprene emissions. It is not clear whether a fraction of  
25 these short lived compounds is oxidized into  $\text{CO}_2$  in the eddy flux tower footprint, and hence included in the  $NEE_{obs}$  flux. If such a large flux  $F_{VOC-C}$  lost by grasslands to the

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atmosphere is confirmed by observation in the future, then it would cancel out most of the NBP sink (see below). Therefore, the uncertainty on  $F_{\text{VOC-C}}$  is a major source of error when deriving NBP from eddy-covariance NEE by Eq. (1).

### 3.6 Net biome productivity

#### 3.6.1 NBP from eddy-covariance sites

Using Eq. (1) at the golden sites of Table 2 gives  $\text{NBP}_{\text{obs}} = 57 \pm 11 \text{ g C m}^{-2} \text{ yr}^{-1}$ . Using Eq. (3) for upscaling gives  $\text{NBP}^*_{\text{obs}} = 74 \pm 14 \text{ g C m}^{-2} \text{ yr}^{-1}$ . It must be noted that this is in any case a very rough extrapolation of NBP, based upon a limited number of sites with rather high productivity/intensive management schemes, and measured over 6 yr at the most. These two NBP estimates are derived as the sum of fluxes of larger absolute values and opposite sign. Hence, systematic errors contained into each of these fluxes and biases induced for instance by neglecting VOC (and possibly fires in southern Europe), affect the NBP calculation. Sporadic plowings will induce C losses later followed by C sinks. Eddy-flux measurements did not include plowing events or other disturbances, implying that this NBP estimate is likely biased towards a sink. At face value, we do not account for possible high-manure returns to grasslands within intensive livestock production systems which produce excess manures from feed compounds such as maize silage, grains, soybean cakes.

#### 3.6.2 Land use change induced NBP

The ORCHIDEE land use change NBP component of grasslands is a small sink of  $2 \text{ g C m}^{-2} \text{ yr}^{-1}$  across EU-27 over 1990–1999. This small flux indicates that the contribution of recent land use change, mainly farmland abandonment, on grassland NBP is very limited, due to the fact that the grassland area remained very stable over the past 20 yr. In total,  $0.0015 \times 10^6 \text{ km}^2$  of grassland disappeared meanwhile  $0.0017 \times 10^6 \text{ km}^2$  were created between 1990 and 2000 (EEA, 2005). In some regions,

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however, changes in farm structure, in EU-agricultural policy, and the transition to market economies in Eastern European countries have been driving larger changes in grasslands. For instance, pasture was created in the Czech Republic following arable land abandonment after 1990, and pasture disappeared in Southern Ireland to be replaced by croplands for animal feed (EEA, 2005).

### 3.7 N<sub>2</sub>O and CH<sub>4</sub> emissions

#### 3.7.1 N<sub>2</sub>O emissions

Results from the three methods: PASIM, the fuzzy logic model, and the disaggregated UNFCCC national statistics (Sect. 3) lead to EU-25 annual N<sub>2</sub>O emissions of 0.25, 0.40±0.05 and 0.29±0.07 g N<sub>2</sub>O m<sup>-2</sup> yr<sup>-1</sup>, respectively. The spread between these estimates defines an uncertainty range of 0.07 g N<sub>2</sub>O m<sup>-2</sup> yr<sup>-1</sup>. Independent measurement of annual N<sub>2</sub>O emissions at a set of 9 sites reached 0.10±0.05 g N<sub>2</sub>O m<sup>-2</sup> yr<sup>-1</sup> (Soussana et al., 2007; Flechard et al., 2007) lower than our European-scale estimates. Based on GWP at 100-yr horizon (IPCC, 1995), direct N<sub>2</sub>O emissions by grassland soils cancel 13% of NBP<sub>obs</sub>. However, reactive N cycling may also contribute to indirect N<sub>2</sub>O emissions which are not counted here. The discrepancy between our three different estimates reflect errors in the amount of N-fertilizer applied or in the emission factor relating N applied to N<sub>2</sub>O emissions per unit area. For instance, manure N-application is missing in the PASIM flux estimate. Consideration of this source will certainly lead to a more accurate estimate. Even if our estimates of N<sub>2</sub>O emissions compare relatively well with each other at EU-25 level, they differ widely at the country level (Figs. 5 and 6). The main discrepancy is observed in Sweden and Finland, where the UNFCCC statistics based estimate is higher than both PASIM and the fuzzy-logic model results. This comes from histosol soils where the GIS soil database used in the UNFCCC statistics assumes a significant grassland area (27%) while the FAO statistics used in the PASIM and fuzzy-logic model indicate a very small grassland area.

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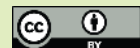
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## 3.7.2 CH<sub>4</sub> emissions

We estimate CH<sub>4</sub> emissions from enteric fermentation to 6.7 Tg CH<sub>4</sub> yr<sup>-1</sup> for EU-25 in 2000. The main country contributors are France (1.3 Tg), Germany (0.9 Tg) and UK (0.8 Tg). In western Europe, concentrate feed accounts for a large fraction of the digestible energy supplied to ruminants, e.g. 125 Mt of corn and of wheat are grown for feed in Europe (FAO, 2006). The fraction of CH<sub>4</sub> emissions from enteric fermentation directly related to grassland NPP is not known, but can be estimated at ca. 50% as a very rough estimate. This leaves some 3.4 Tg CH<sub>4</sub> emissions derived directly from grassland grazing at EU-25-scale. Dividing this flux by the grassland area considered in this study leads to a flux density of 5.8 g CH<sub>4</sub> m<sup>-2</sup> yr<sup>-1</sup>. Expressed as CO<sub>2</sub>-equivalents (IPCC, 1995), CH<sub>4</sub> emissions from pasture thus cancel 63% of NBP. A 40% uncertainty is associated to these CH<sub>4</sub> emissions, as we hardly know the share of CH<sub>4</sub> coming from grazed grass vs. other sources of feed.

## 4 Discussion

### 4.1 Net primary productivity uncertainties and spatial gradients

This discussion section is focused on the uncertainties associated to grassland GHG fluxes, and on comparison with former studies. It is difficult to quantify grassland NPP uncertainties, because observations at representative sites are mostly lacking, and NPP is virtually impossible to measure (Lauenroth et al., 2006; Scurlock et al., 2002). Obviously, our estimate of NPP<sub>obs</sub><sup>\*</sup> in this study is very uncertain because it is based on few sites. Yet, NPP<sub>obs</sub><sup>\*</sup> lies in good agreement with NPP<sub>PASIM</sub> calculated at 1° by 1° across Europe, given the model error of 20% (Sect. 3). This result is encouraging for the prospect to better constrain grassland NPP using eddy covariance GPP measurements and independent measurement (labeling experiments) of the GPP to NPP ratio.

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The  $NPP_{PASIM}$  carries the following sources of uncertainties: 1) the amount of N fertilizer applied to each grid point is not known with sufficient precision, 2) the fact that real world grasslands are exploited simultaneously for cutting and grazing, whereas the PASIM model makes an artificial distinction between these two practices, implying the simulated difference between pasture and meadows NPP to be over-estimated, and 3) the fact that the management rules developed to run PASIM all grasslands to be managed for sustaining the highest possible number of animals, an assumption that leads to a high bias of NPP.  $NPP_{PASIM}$  better represents intensively managed “green carpets” of regions with high fertilizer inputs and year-round growing conditions such as Ireland, Belgium, the Netherlands, Western France and Northern Spain, and is less accurate for of less productive and dryer grassland systems in Mediterranean and Eastern European countries.

Gilmanov et al. (2007) observed a gradient of GPP in grasslands across Europe, going from  $1800 \text{ g C m}^{-2} \text{ yr}^{-1}$  in Ireland down to  $460 \text{ g C m}^{-2} \text{ yr}^{-1}$  in the dryer Eastern Hungary. This gradient is caused by rainfall and by a shorter growing season at more continental sites. Because NPP broadly scales with GPP, we also expect a 3-fold decrease in NPP along a West-to-East gradient, and along a Temperate-to-Mediterranean dryness gradient. In Fig. 4, such a spatial gradient in  $NPP_{PASIM}$  seems to be captured well.  $NPP_{PASIM}$  decreases from  $1300 \text{ g C m}^{-2} \text{ yr}^{-1}$  down to  $800 \text{ g C m}^{-2} \text{ yr}^{-1}$  between Western and Eastern Europe, and from  $1000 \text{ g C m}^{-2} \text{ yr}^{-1}$  to  $700 \text{ g C m}^{-2} \text{ yr}^{-1}$  between Temperate and Southern Europe (Fig. 4d). In summary, our estimate of EU-25 grassland NPP is rather high ( $750$  to  $797 \text{ g C m}^{-2} \text{ yr}^{-1}$ ) typically 20% higher than cropland NPP (Ciais et al., 2010) and 70% higher than forest NPP (Luyssaert et al., 2010). This is not so surprising since grasslands have a high LAI (Owen et al., 2007), can sustain photosynthesis all year round in suitable climates (Gilmanov et al., 2007) and receive N-inputs by fertilizers while being able to maintain a much faster N recycling than forests, in particular for pastures.

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## 4.2 Net ecosystem exchange in relation to NBP

The flux of carbon exported away from the ecosystem where NEE is measured (cut and grazed herbage, VOC emissions, and DOC leaching to rivers) is usually greater than the flux of external carbon input from organic fertilizers. The mean input of C from manure spreading is  $48 \text{ g C m}^{-2} \text{ yr}^{-1}$  over EU-25, inferred to balance the numbers in Table 3. This is close to  $30 \text{ g C m}^{-2} \text{ yr}^{-1}$  of manure spreading reported from agricultural statistics (Smith, unpublished data). Therefore, NBP is expected to be a significantly smaller carbon sink than NEE. Our NBP estimate of  $57 \pm 143 \text{ g C m}^{-2} \text{ yr}^{-1}$  confirms this hypothesis, giving an NBP to NEE ratio of 23% averaged from the 11 sites. This confirms that NBP of grassland systems cannot be simply analyzed from eddy-covariance assets alone. A large part of harvested carbon not measured by NEE (roughly 30% of NPP after Sect. 2) accumulates into forage digested by herbivores in farm buildings and released as  $\text{CO}_2$  outside of the footprint measured by the eddy covariance instrumentation. Because of this compensatory source of  $\text{CO}_2$ , the net grassland  $\text{CO}_2$  sink at larger spatial scale will be closer to NBP than to NEE.

## 4.3 Net biome productivity

### 4.3.1 Comparison with former studies

Janssens et al. (2003) provided an estimate of  $\text{NBP} = 66 \pm 90 \text{ g C m}^{-2} \text{ yr}^{-1}$  over  $1.5 \times 10^6 \text{ km}^2$  for geographic Europe grasslands. Our estimate is comparable to their value, but it has the advantage to rely at least partially on measurements. Janssens et al. (2003) derived NBP by combining the CESAR model projections for 2008–2012 (Vleeshouwers and Verhagen, 2002) with land-use data for the period 1990–1995. The CESAR model was developed for modeling climate change impacts, and not to determine the baseline carbon flux related to management and climate. CESAR estimates were based on country scale yield statistics, considering that an arbitrary fraction of 44% of NPP was harvested and so exported. Yields were prescribed at

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a national scale, the harvested flux  $H$  was fixed over all Europe, and the impact of grazing was very crude. The CESAR model uncertainties are therefore virtually unquantified. Our NBP estimate is a larger sink than the European-wide simulation results obtained with the Roth-C soil carbon model (Smith et al., 2005, see Fig. 3). Smith et al. (2005) estimated that European grassland soils in EU-25 plus Norway and Switzerland ( $0.63 \times 10^6 \text{ km}^2$ ) were a net sink in the period 1990–1999 of  $36 \pm 18 \text{ g C m}^{-2} \text{ yr}^{-1}$ .

### 4.3.2 Comparison with regional soil C inventories

Our grassland NBP estimate is a larger sink than SOC accumulation observed by regional soil inventories in Belgium, France and UK (Bellamy et al., 2005; Thompson et al., 2005; Howard et al., 1995). Inventory-based NBP is summarized in Table 2. For instance, Bellamy et al. (2005) inferred SOC changes from the UK national soil C inventory with a 5 km grid sampling, which constrains regional NBP to a *source* in the range 2 to  $35 \text{ g C m}^{-2} \text{ yr}^{-1}$ . On the other hand, Smith et al. (2007) indicated that soil bulk density may not have been estimated correctly, so that Bellamy et al. (2005) may have overestimated SOC losses. A recent study of two long-term grassland experiments in UK concluded to no significant variation in SOC stocks due to climate trends, when the management was kept constant (Hopkins et al., 2009).

A source of discrepancy between NBP derived from regional SOC inventories and NBP derived from eddy-covariance NEE as in our study, is obviously the amount of carbon exported by DOC leaching and VOC. This flux is not assessed from eddy-covariance data, but is implicitly included in inventory measurements of SOC changes. The export of DOC alone can represent up to 8% of annual NEE (Don et al., 2008). We do not know accurately the magnitude of leached DOC flux at each site, hence neither its corresponding fraction of NEE. In addition, we do not know if leached DOC comes from recent assimilates or from very old carbon pools. We used by default in our NBP calculation a total export to rivers of  $9.6 \text{ g C m}^{-2} \text{ yr}^{-1}$  identical at each site, which represents 4% of annual NEE averaged across the 11 sites. This value given by the mean export of European rivers is probably underestimated for northern European

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grasslands with high SOC content. Our estimate of VOC losses to atmosphere from a model also suggest that this flux is large and may cancel out most of the NBP sink if NBP is derived from NEE by Eq. (1). On the other hand, SOC inventories are prone to systematic bias of NBP towards underestimation of sink. Typically only the top soil is being surveyed for agricultural land, while carbon stock and carbon accumulation in deeper soil horizons are ignored. Deep SOC stocks can be large for grasslands (Fontaine et al., 2007). In general, grassland NBP would be best addressed with large-scale and long-term soil carbon stock inventories with sufficient number of replicate samples and vertical profile information. Unfortunately, these are currently not available across Europe.

In summary, our new estimate of EU-25 grassland NBP is similar to the former study of Janssens et al. (2003) and 1.8 times higher than the model of Smith et al. (2005) but uncertainties overlap. Our estimate is derived primarily from data, although upscaling from few sites to Europe has a large uncertainty. In particular, the sites at which information on input and export are available for this study to deduce NBP are biased towards intensive management, which may imply an overestimated NBP sink over the EU-25. Estimates of carbon exports at the eddycovariance sites including VOC and DOC, would be needed to avoid systematic sink overestimation. To improve future model estimates, one would need to drive PASIM or another grassland carbon model, at continental scale with changing climate and agricultural technology, as it was done for croplands using three models (see Ciais et al., 2010 for instance). Such a simulation of NBP would require accurate reconstruction of management drivers, in particular of grassland plowing frequency, which is expected to make soil carbon sequestration at regional scale lower than currently modeled.

#### 4.4 Carbon sequestration efficiency

We define the Carbon Sequestration Efficiency (CSE) by the ratio NBP/NPP. The CSE of European grasslands is estimated at 0.09 using  $NPP_{PASIM}$ , and at 0.11 using  $NPP_{obs}^*$ . The full CSE uncertainty range is 0.05 to 0.18 when using different NBP and NPP

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estimates from this study. This makes grasslands good candidate ecosystems for maintaining carbon sinks. Management options that increase NPP such as fertilization, irrigation, inter-sowing of grasses and legumes will have the potential to increase soil organic carbon (Conant et al., 2001). However, most grassland systems in Europe are the result of forest clearing and will therefore have to be maintained by continuous management. Nutrients are also needed to store additional carbon (soil organic matter has a lower C:N ratio than litter inputs) such that the current carbon sink needs to be matched by N inputs from either N fertilizers or biological N fixation. Studies on grasslands with little management found NBP ranging from nearly zero to a small sink (Novick et al., 2004; Don et al., 2009). Very high management intensity decreases NBP and increases N<sub>2</sub>O and CH<sub>4</sub> emissions (Soussana et al., 2007), which suggests that for enhancing carbon sequestration an optimum grassland management intensity may exist.

A second remark regarding the high CSE of European grasslands is the distinction between pasture and meadows. In the former, both NPP is higher *and* removals are lower than in the latter, hence favoring carbon sequestration in soils. This expected NBP difference between pasture and meadows is very uncertain, as there are still insufficient sites to test it in the field. Further, with both practices being usually used alongside each other at the farm scale, theoretical differences between grazed and cut grasslands NBP will get smoothed at the farm-scale and regional scale.

The CSE of grasslands is (slightly) lower than the CSE of forests equalling 0.14 (Luyssaert et al., 2010). Such an apparently lower ability of grasslands to store carbon for a given NPP must be tempered by two remarks. Firstly, in the presence of periodic wood biomass removal in forests, and of permanent grass biomass removal (grazing, mowing) in grasslands, it is more relevant to compare ratios of NBP to carbon input to the soil, rather than ratios of NBP to NPP. The soil C input to NPP ratio is 0.1 for grasslands against 0.03 for forests (from Luyssaert et al., 2010). For a given soil carbon input, grasslands thus seem capable of SOC sequestration 3 to 4 times *more efficiently* than forests. However, forests allocate roughly 1/3 of their NPP to wood

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growth, which grasslands obviously can't. This woody biomass sink, if not reduced by timber harvest, give a significant carbon sink advantage to forests, despite their lower soil carbon storage.

#### 4.5 CH<sub>4</sub> and N<sub>2</sub>O emissions offsetting carbon sinks

5 CH<sub>4</sub> and N<sub>2</sub>O emissions by grassland soils counterbalance 70 to 80% of the European grassland carbon sink (NBP<sub>obs</sub>). Our new estimate gives a higher role for non-CO<sub>2</sub> gases offsetting carbon sink, than the local measurements from 9 grassland sites reported in Soussana et al. (2007). The latter study indicated N<sub>2</sub>O and CH<sub>4</sub> emissions offsetting only 44% of NBP. More sites and measurement years will be needed to reduce the uncertainty, which is still substantial, in this trade-off between carbon storage and non-CO<sub>2</sub> trace gas emissions by grasslands. N<sub>2</sub>O emissions are roughly nine times higher for grazing than for cutting. CH<sub>4</sub> animal emissions also should be added to the greenhouse budget of pasture, and indirectly of meadows, since the latter provide forage digested indoors leading to further enteric fermentation (Soussana et al., 15 2009). Model estimates of N<sub>2</sub>O and CH<sub>4</sub> emissions are uncertain. For instance, if more N-fertilizer is prescribed to the PASIM model, NPP will increase but the NPP use efficiency (H/NPP) will also increase, thereby decreasing carbon input to the soil and NBP. Adding N fertilizers could increase carbon sequestration by increasing GPP, but resulting in increased N<sub>2</sub>O and CH<sub>4</sub> emission fluxes. In the study of Vuichard et al. (2007b), 20 non-CO<sub>2</sub> gases emissions offset 28% of the potential C sequestration under current N-fertilization rates and could offset up 50% of this potential if fertilization rates increase. Vuichard et al. did not include indirect N<sub>2</sub>O emissions from N leached or volatilised as NH<sub>3</sub>, which would even further increase the offsetting effect of N<sub>2</sub>O emissions.

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## 5 Conclusions

The main findings of this synthesis of European grassland carbon fluxes are:

- Based upon site observations, the carbon balance of EU-25 grasslands is a carbon sink ranging from 57 to 74 g C m<sup>-2</sup> yr<sup>-1</sup> according to the method used for calculation. NBP estimated from averaging data at the few eddy covariance where NBP can be estimated from NEE and lateral import/export is consistent with NBP up-scaled using the GPP spatial distribution of the PASIM model.
- NBP of grasslands estimated in this study from CARBOEUROPE data is similar to the former model estimate of Janssens et al. (2003). The limited number of sites, their short measurement period (1 to 6 yr) and similar intensive management characteristics are major source of uncertainty, likely resulting in overestimating the C sink.
- NPP of grasslands is quite high (750 to 797 g C m<sup>-2</sup> yr<sup>-1</sup>) that is 20% higher than cropland and 70% higher than forest NPP. This can be explained by on average intensive management conditions and favorable climate.
- Grasslands emit N<sub>2</sub>O and CH<sub>4</sub>, which are estimated in this study to offset 13±5% and 63±25% of the carbon sink, when using a 100 yr GWP. The uncertainty comes from three different methods in the case of N<sub>2</sub>O emissions, and from an assumption on the share coming from grasslands vs. other sources of feed in the case of CH<sub>4</sub> emissions.

Future studies of the grassland GHG balance will need to rely upon a larger number of sites and longer time series of observations. Uncertainties are still large in this work, but are expected to be reduced in the future, with more data being collected. In this context, it will be important to verify NBP scaled up from site-observations with information on farm-level practice and satellite observation, with independent SOC inventories. Specific measurements of the fraction of NEE lost to rivers by DOC export and lost to

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the atmosphere by VOC emissions will also be important to improve the accuracy of NBP derived from eddy covariance NEE.

Modelers have paid so far too little attention to grasslands. Models capable to simulate nitrogen cycling and the specifics of management in grasslands, but also disturbances such as plowing, are needed. In parallel, geospatial information on the intensity of management practice must be collected to drive these models. In the longer run, the future evolution of the grassland GHG balance in Europe, in particular its vulnerability to soil warming and its sustainability under a range of adaptation management scenarios could be predicted by models calibrated by available data, for instance using data assimilation techniques.

## Appendix A

### Component of the carbon balance use in all the manuscripts

Gross primary production (GPP) of an ecosystem represents the gross uptake of CO<sub>2</sub> that is used in photosynthesis. The synthesis of new plant tissues, water and nutrients and the maintenance of living tissues require energy, which is produced during autotrophic respiration and emits CO<sub>2</sub> to the atmosphere ( $R_a$ ). The latter process consumes roughly half of the photo-assimilates (Luyssaert et al., 2007). The quantity of photosynthates not used for respiration and therefore available for other processes is defined as net primary production (NPP) and relates to GPP and  $R_a$  as:

$$GPP = NPP + R_a \quad (A1)$$

To date, direct measurements of total NPP are impossible. Part of the organic material produced during NPP is lost via emission of Volatile Organic Compounds (VOC), exudation from roots or carbon transfer to root symbionts. Although the bulk of NPP is allocated to the production of above- and below-ground biomass, this fraction is also difficult to assess. Firstly, not all of the biomass produced remains on site until the point at which the measurements are made, and corrections need to be made for

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this removed biomass. Examples of biomass removal processes include harvest and herbivory, both particularly important in managed grasslands, as well as the fast decomposition of senescent plant material (fine roots and leaves turnover). The turnover of fine root and leaves can be extremely difficult to assess, especially in grassland and in evergreen forests. Hence, NPP estimates are always highly uncertain (Lauenroth et al., 2006; Scurlock et al., 2002). Depending on the ecosystem NPP is typically estimated as:

$$\text{NPP} = \text{NPP}_{\text{foliage}} + \text{NPP}_{\text{wood}} + \text{NPP}_{\text{roots}} + \text{NPP}_{\text{residual}} \text{ in forest} \quad (\text{A2a})$$

$$\text{NPP} = \text{NPP}_{\text{foliage}} + \text{NPP}_{\text{roots}} + \text{NPP}_{\text{grazing}} + \text{NPP}_{\text{cutting}} + \text{NPP}_{\text{residual}} \text{ in grasslands} \quad (\text{A2b})$$

$$\text{NPP} = \text{NPP}_{\text{foliage}} + \text{NPP}_{\text{seeds/fruits}} + \text{NPP}_{\text{roots}} + \text{NPP}_{\text{residual}} \text{ in croplands} \quad (\text{A2c})$$

In all the manuscripts, all but the latter terms are included in NPP estimates, except  $\text{NPP}_{\text{residual}}$  (VOC, rhizodeposition, unintended grazing by insects and game). In addition to the high uncertainty in the NPP estimates, this  $\text{NPP}_{\text{residual}}$  term also implies a systematic underestimation of all reported NPP's.

Each year, part of the biomass produced is transferred to litter- and/or soil layer carbon pools (each of which has different residence times). These carbon pools are subject to decomposition, a process defined as heterotrophic respiration ( $R_h$ ). Decomposition processes that contribute to  $R_h$  include decomposition of biomass from the current year, but also decomposition of organic matter that accumulated in the ecosystem during the last decades, centuries or millennia. The short-term imbalance between NPP and  $R_h$  is termed Net Ecosystem Productivity (NEP):

$$\text{NEP} = \text{NPP} - R_h \quad (\text{A3})$$

The sum of  $R_h$  and  $R_a$  represents the total ecosystem respiration ( $R_e$ ). The sum of the belowground fraction of  $R_a$  and  $R_h$  is the soil respiration. NEP is also defined by the short-term difference between GPP and  $R_e$ :

$$\text{NEP} = \text{GPP} - R_e \quad (\text{A4})$$

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A flux that can be measured by eddy-covariance technique is the net CO<sub>2</sub> exchange between ecosystem and atmosphere, or NEE. By convention carbon gain by ecosystems defines negative NEE. However, NEE observed at an eddy covariance site differs from the long-term carbon balance because non-CO<sub>2</sub> carbon losses and non-respiratory CO<sub>2</sub> losses, which occur at a range of time scales, are typically ignored. Shortly (<1 yr) after uptake, synthesized compounds are lost from the ecosystem as VOC, which defines a carbon flux  $F_{\text{VOC}}$ . On longer time scales (>1 yr), part of the annually accumulated NEE leaves the ecosystem to ground water as dissolved, particulate organic or inorganic carbon, which defines a flux leached to rivers  $D$ . Carbon can also be lost to the atmosphere as CH<sub>4</sub> produced from wet soils. In pastures, additional CH<sub>4</sub> losses occur from grazing animals enteric fermentation, which defines a flux  $F_{\text{CH}_4}$ . Finally, part of the carbon that has been built up over the years into ecosystems by NEE leaves the ecosystem and eventually returns to the atmosphere by fires ( $F$ ), by cut or grazing “harvesting” ( $H$ ), and by erosion/re-deposition processes ( $E$ ).

In grasslands and croplands organic carbon can enter the ecosystem through fertilization with manure or slurry ( $I$ ), and through re-deposition of eroded sediments and deposition of dissolved carbon. In pastures, carbon embedded into the flux  $F_{\text{CH}_4}$  is also included in the “harvest”  $H$ . To avoid double-accounting, we separate  $H$  between a grazing and cutting flux  $(1-\alpha)H$  returned to atmosphere as CO<sub>2</sub>, and a grazing flux  $\alpha H$  emitted as CH<sub>4</sub> by animals. If the grazing animals only feed from the pasture, then  $F_{\text{CH}_4} = \alpha H$  in carbon mass. If grazing animals receive complementary diet, then  $F_{\text{CH}_4} > \alpha H$ . Animal products derived from grassland biomass, such as milk and liveweight gain also remove carbon from grasslands (see Soussana et al., 2007). This flux is however included into the exported grazing and cutting flux  $(1-\alpha)H$ . All the non-CO<sub>2</sub> carbon losses and non-respiratory carbon losses and gains should be accounted for to obtain the carbon-balance. The Net Biome Productivity NBP (Schulze und Heimann, 1998; Buchmann und Schulze, 1999; Chapin et al., 2005) is the term applied to the total rate of organic carbon accumulation (or loss) from ecosystems. NBP is positive when

ecosystems gain carbon, and can be calculated using Eq. (A5)

$$\text{NBP} = -\text{NEE} + I - (1 - \alpha) \cdot H - F_{\text{CH}_4\text{-C}} - F_{\text{VOC-C}} - E - D - F_{\text{fires}} \quad (\text{A5})$$

Although Eqs. (A1) to (A5) are based on solid theoretical principles, methodological issues remain unsolved when putting them to practice. Gross primary productivity estimates derived from eddy covariance measurements do not always account for light induced inhibition of foliar respiration (see e.g., Gilmanov et al., 2007) and are sensitive to biased NEE estimates due e.g. to advection. The consequences for GPP estimates are still under debate (Pinelli and Loreto, 2003; Wohlfahrt et al., 2005). NEE measurements by eddy covariance techniques suffer from incoming or outgoing advective fluxes, which could result in over- or under-estimation of NEE (Feigenwinter et al., 2008).

On the other hand, NPP is typically underestimated by any measurement method because only its major biomass components are measured, whereas its minor components ignored (Eq. 2; Clarck et al., 2001; Scurlock et al., 2002). Depending on the methodology applied to estimate  $R_h$ , respiration from mycorrhizae is considered as auto- or heterotrophic respiration (Högberg et al., 2001). Despite these methodological shortcomings, independent measurements of the different components are often consistent, indicating that these issues have either a small effect or are compensating each other (Luyssaert et al., 2010).

## Appendix B

### Abbreviations used in all the manuscripts

**CAPRI DynaSpat:** EU project concerning the Common Agricultural Policy Regional Impact Assessment – The Dynamic and Spatial Dimension

**CSE:** carbon sequestration efficiency defined as the ratio of NBP to NPP

**D:** C flux due to DOC losses from soils to rivers and lakes ( $\text{g}^{-2} \text{yr}^{-1}$ )

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**DIC:** dissolved inorganic carbon

**DOC:** dissolved organic carbon

**E:** C flux due to soil C erosion ( $\text{g}^{-2} \text{yr}^{-1}$ )

**EU-25:** European Union with its 25 member state (31.12.2006): Austria, Belgium, Cyprus, Czech Republic, Denmark, Estonia, Finland, France, Germany, Greece, Hungary, Ireland, Italy, Latvia, Lithuania, Luxembourg, Malta, Netherlands, Poland, Portugal, Slovakia, Slovenia, Spain, Sweden and UK. Thus excluding Belarus, European Russia, Turkey and Ukraine.

**F:** C flux due to forest fires ( $\text{g C m}^{-2} \text{yr}^{-1}$ )

**GHG:** greenhouse gas

**H:** C flux due to harvest ( $\text{g C m}^{-2} \text{yr}^{-1}$ )

**I:** C flux due to manure application ( $\text{g m}^{-2} \text{yr}^{-1}$ )

**LUCAS:** Land Use and Cover Area Frame Statistical Survey

**NBP:** net biome production ( $\text{g C m}^{-2} \text{yr}^{-1}$ )

**NCEP:** National [US] centre of environmental protection

**NEE:** net ecosystem exchange ( $\text{g C m}^{-2} \text{yr}^{-1}$ )

**NPP:** net primary production ( $\text{g C m}^{-2} \text{yr}^{-1}$ )

**NUTS:** Nomenclature of Territorial Units for Statistics, is a geocode standard for referencing the administrative divisions of countries for statistical purposes. NUTS-3 is the most detailed level and divides the EU-25 in 1243 administrative units. NUTS-2 divides the EU-25 in 257 administrative units.

**PASIM:** process-oriented model specifically designed for managed grassland ecosystems

**POC:** particulate organic carbon

**$R_a$ :** autotrophic respiration ( $\text{g C m}^{-2} \text{yr}^{-1}$ )

**$R_e$ :** total ecosystem respiration ( $\text{g C m}^{-2} \text{yr}^{-1}$ )

**$R_h$ :** heterotrophic respiration ( $\text{g C m}^{-2} \text{yr}^{-1}$ ). For Croplands  $R_h$  is divided in  $R_{h1}$  and  $R_{h2}$ .  $R_{h1}$  is the heterotrophic respiration on-site where  $R_{h2}$  denotes the off-site heterotrophic respiration of harvest products

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**SOC:** soil organic carbon

**UNFCCC:** United Nations Framework Convention on Climate Change

**VOC:** volatile organic carbon

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**Table 1.** Data sources of the analyses presented in this study. The data sources and or model structures are described in detail in tabulated references.

	Site-observation	Models	Inventories
Net primary production (NPP)	Inferred from the initial slope of new carbon input to the soil in one <sup>13</sup> C labeling experiment (Klumpp et al., 2007)	PASIM with the ratio of grazed to cut grasslands being defined from simple rules (Vuichard et al., 2007a,b) PASIM output corrected with the ratio of grazed to cut grasslands defined by regional statistics on <i>extensive</i> and <i>extensive</i> management (this study)	
Harvest ( <i>H</i> )	Calculated animal intake and forage grass harvesting at 9 sites (Soussana et al., 2007)	Estimated from modeled management rules (cutting≈each 6 weeks and grazing during growing season) in Vuichard et al., 2007a,b	
Fires ( <i>F</i> )		MODIS products (Giglio et al., 2006) coupled to the fire module of the CASA model (van der Werf et al., 2006)	
Losses to rivers ( <i>D</i> )			European rivers database (Meybeck et al., 2005) up-scaled on the basis of runoff, land cover and rock types similarities (Ciais et al., 2008)
Heterotrophic respiration (HR)		PASIM+mass balance calculation	
Net biome production (NBP)	Eddy covariance NEE corrected with manure input (I) and forage exports (E) of forage from 11 sites (Table 2)	PASIM simulated GPP multiplied by observed NBP/GPP ratio	Soil carbon inventories (limited regional data)

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**Table 2.** Literature survey of NBP estimates at various European grassland sites, and from the national inventories. A positive NBP denotes a net carbon accumulation in grassland ecosystems. All fluxes are in  $\text{g C m}^{-2} \text{yr}^{-1}$ .

	Region	Site Name	NEE	NBP	Harvest	Remarks	Measurement period [mon]	Precipitation [mm]	Elevation [m a.s.l.]	Method	Source
<b>Ecological site studies</b>											
extensive pasture	Hungary	Bugac	69	68	0		24	500	140	eddy covariance	Soussana et al., 2007
extensive pasture	Germany	?	66	7	59		33	547	290	eddy covariance	Don et al. in review
extensive pasture	Italy	Malga Arcapo	360	358	0		12	1200	1699	eddy covariance	Soussana et al., 2007
extensive pasture	France	Laqueuille	75	69	1	Extensively grazed pasture without N application	36	1200	1040	eddy covariance	Allard et al., 2007
intensive pasture	France	Laqueuille	99	87	2	Intensively grazed permanent pasture with N application	36	1200	1040	eddy covariance	Allard et al., 2007
extensive meadow	Swiss	Oensingen	254	-57	311	Sown grass-clover mixture without N fertilizer	36	1100	450	eddy covariance	Ammann et al., 2007
intensive meadow	Swiss	Oensingen	467	147	368	Sown grass-clover mixture with N fertilizer	36	1100	450	eddy covariance	Ammann et al., 2007
intensive grassland	Netherlands	Lelystad	177	33	220		24	780	-5	eddy covariance	Soussana et al., 2007
intensive grassland	Scotland	Eastar Bush	343	231	110		24	638	190	eddy covariance	Soussana et al., 2007
intensive grassland	Ireland	Carlow	293	-170	374		24	824	56	eddy covariance	Soussana et al., 2007
intensive grassland	UK	Somerset Levels	169	-34	228	Wet grassland; no account for $\text{CH}_4$ ; corrected for animal intake ( $25 \text{ g C m}^{-2} \text{ yr}^{-1}$ )	12	750		eddy covariance	Lloyd 2006
intensive grassland **	Ireland	Cork county	15	15	0	Newly established; the amount of manure/slurry applied is not reported	12	1470	180	chamber+ aboveground cutting every 4 weeks	Byrne et al., 2005
intensive grassland **	Ireland	Cork county	38	38	0	Permanent grassland; the amount of manure/slurry applied is not reported	12	1470	180	chamber+ aboveground cutting every 4 weeks	Byrne et al., 2005
intensive meadow **	Denmark	Lille Valby	152	1277	333	Received massive application of manure during measurement period	24	731	15	eddy covariance	Soussana et al., 2007
<b>inventories</b>											
grassland	Belgium			22 to 44		national inventories	40 yr			soil C stock changes 0–100 cm	Lettenens et al., 2005
grassland	England, Wales			-2 to -38		national inventories	25 yr			soil C concentration changes 0–15 cm	Gojts and van Wesemael, 2007
grassland	France			20±40 (median)		Chrono-sequences (after land-use change)	10 yr			soil C stock changes 0–30 cm	Soussana et al., 2004
<b>Mean excl. sites (**)</b>			<b>216</b>	<b>67</b>							
<b>Standard error</b>			<b>137</b>	<b>143</b>							

\*\* Sites excluded from the mean.



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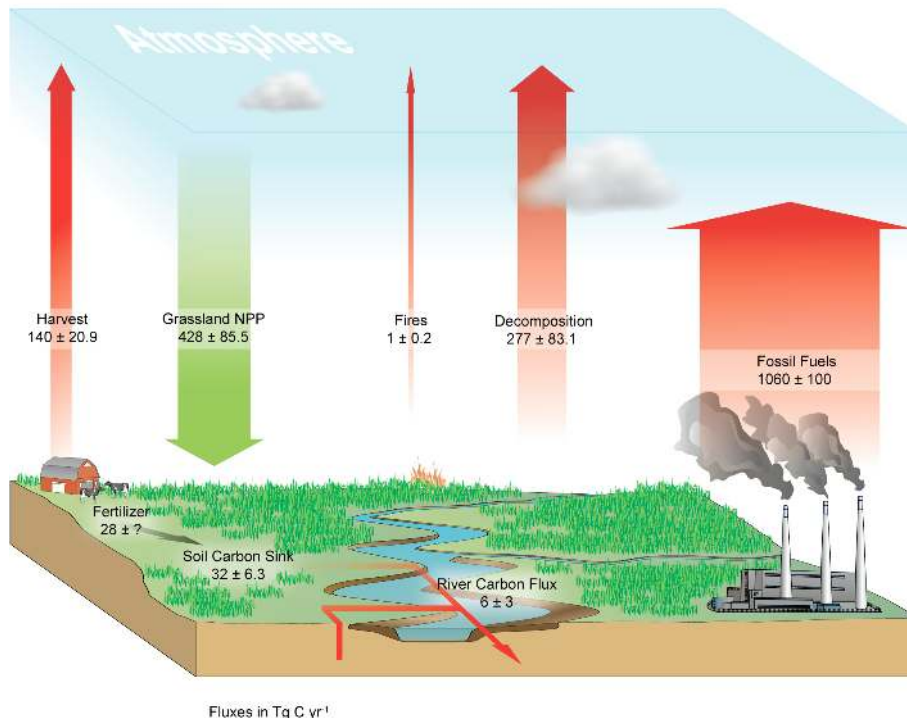


**Table 3.** Summary of the different methods and their estimate used to determine the component fluxes of the European grassland carbon balance.

Method	NPP	$R_h$	NEP	$H$ (exported biomass)	F	$D$	Manure	NBP	NBP
Inventory									
National inventories									26±18
Soil losses to rivers						9.6			
Ecological sites									
FAO network (forage production at 12 sites)	1015±40								
Eddy covariance	758								
GPP×(NPP/GPP) ratio of PASIM model									
Eddy covariance	588								
GPP×(NPP/GPP) ratio from a C13 marking experiment									
Carboeurope sites (average of 11 sites retained in Table 2)			240±70				48.5	67±143	57±11
Ecosystem models									
CASA					0.3				
PASIM pasture (grazed)	1237			218					
PASIM meadow (cut)	646			245					
PASIM total	750±150	486		245*				101±20	91±20

\* Grazed biomass is respired within the ecosystem and is not considered as part of exported flux.

# Grassland



**Fig. 1.** Summary of EU-25 grasslands carbon cycle. Fluxes in Tg C yr<sup>-1</sup> over a grassland area of  $0.57 \times 10^6$  km<sup>2</sup>. Heterotrophic respiration was calculated as the residual term to make the balance close. Fossil fuel emission is the largest single term of the EU25 carbon balance; a companion paper is referring to fossil fuel emissions specifically. Here this flux is shown for comparison with grassland ecosystem fluxes.

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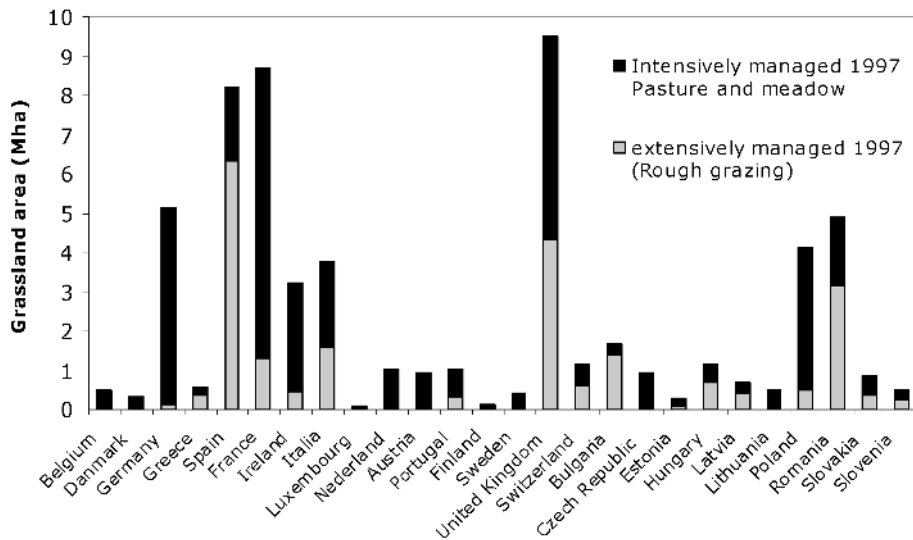
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**Fig. 2.** Extensively and intensively managed grassland area in each EU-25 country, data used to correct simulated NPP by accounting for extensive management. Data are from regional-level EUROSTAT statistics in EU-15, and from national-level FAOSTAT statistics elsewhere.

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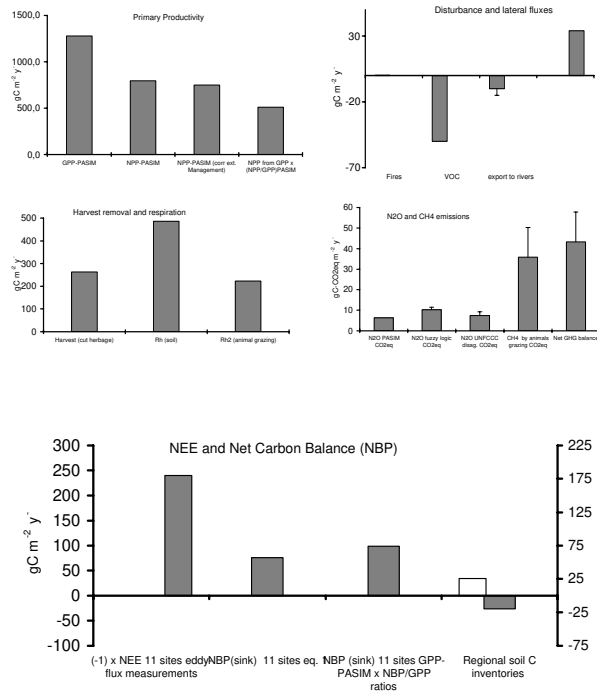
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**Fig. 3.** New estimates of carbon flux components of the EU-25 grasslands, estimated using data and models, and of N<sub>2</sub>O and CH<sub>4</sub> emissions.

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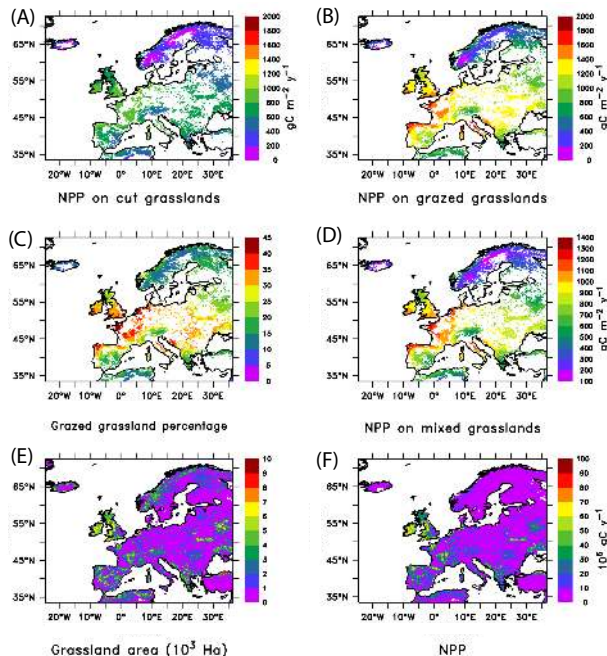
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**Fig. 4.** (A) Distribution of grassland NPP for cut grasslands after the PASIM model. White areas denote grid points with less than 20% grassland land cover. (B) Same as above but for grazed grasslands (pasture). (C) Percentage of pasture area vs. total grassland area, that is pasture plus meadows, in each grid point, as calculated by management rules introduced in the model. (D) Grassland area-weighted NPP with pasture and meadows areas given by C. (E) Grassland area in each  $1 \times 1^\circ$  grid point. (F) Corresponding grassland NPP in  $\text{gC yr}^{-1}$ .

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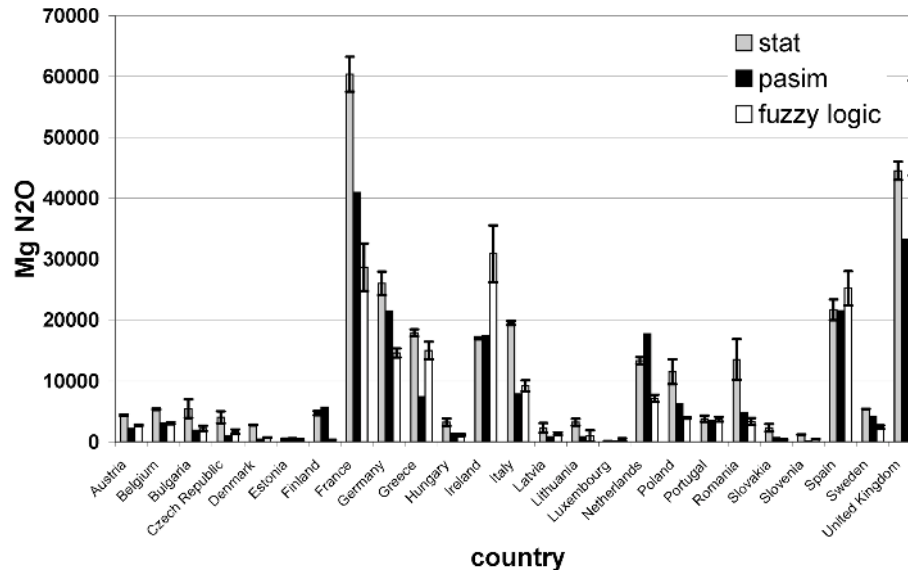
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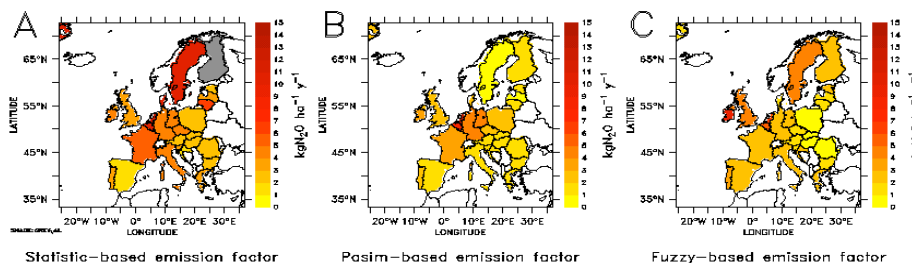
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**Fig. 5.** N<sub>2</sub>O emissions by grassland soils across EU-25 countries, estimated by three methods. Stat=national statistics reported by each country to the UNFCCC, and further apportioned between grasslands and croplands using information on land cover and on percent of fertilizers given to grasslands vs. croplands (Sect. 3). PASIM=spatially explicit simulations of the PASIM process based model combining cut grasslands and pastures. Fuzzy logic=fuzzy logic model developed by Dechow et al. (unpublished data).

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**Fig. 6.** Map of N<sub>2</sub>O fluxes from grasslands averaged in each country of the EU-25+Norway and Switzerland, using 3 different methods. The method based upon statistics (A) gives annual mean values at the scale of each country. The other two methods (B, C) simulate N<sub>2</sub>O fluxes in a spatially explicit and temporally-resolved manner, but the output have been averaged in the figure to be compared with (A).

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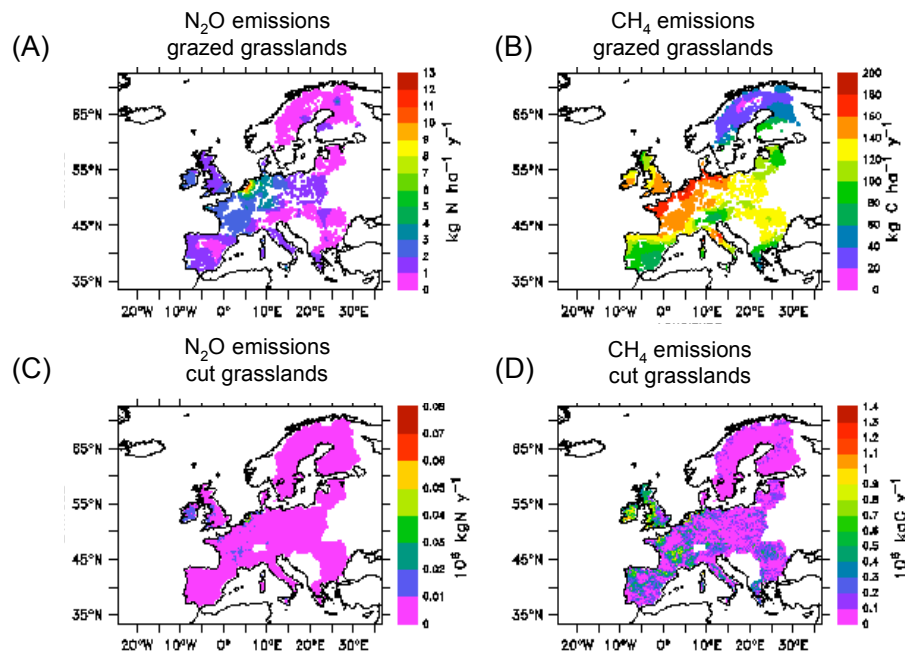
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**Fig. 7.** (A) Distribution of N<sub>2</sub>O fluxes over grassland soils in EU-25 countries after Vuichard et al. (2007b). White areas denote grid points with less than 20% grassland cover. N<sub>2</sub>O emissions are modeled by PASIM at 1×1° from nitrogen cycling processes, fertilizer application, excluding manure, but including animal dung input to pasture. (B) Same but N<sub>2</sub>O emissions are expressed in kg N yr<sup>-1</sup>, showing the distribution of N<sub>2</sub>O emitted to the atmosphere as used for instance in atmospheric transport modelling. (C, D) same but CH<sub>4</sub> emissions by animals grazing on pastures.

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