Effect of climate change, CO2 trends, nitrogen addition, and land-cover and management intensity changes on the carbon balance of European grasslands

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Running head: Attribution the European grasslands NBP trend

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Abstract

Several lines of evidence point to European managed grassland ecosystems being a sink of carbon. In this study, we apply ORCHIDEE-GM a process-based carbon cycle model that describes specific management practices of pastures and the dynamics of carbon cycling in response to changes in climatic and biogeochemical drivers. The model is used to simulate changes in the carbon balance (i.e., Net Biome Production, NBP) of European grasslands over 1991-2010 on a 25 km × 25 km grid. The modeled average trend of NBP is 1.8 - 2.0 g C m^{-2} yr^{-2} during the past two decades. Attribution of this trend suggests management intensity as the dominant driver explaining NBP trends in the model (36% - 43% of the trend due to all drivers). A major change in grassland management intensity has occurred across Europe resulting from reduced livestock numbers. This change has 'inadvertently' enhanced soil C sequestration and reduced N_2O and CH_4 emissions by 1.2 - 1.5 Gt CO_2-equivalent, offsetting more than 7% of greenhouse gas emissions in the whole European agricultural sector during the period 1991-2010. Land-cover change, climate change and rising CO_2 also make positive and moderate contributions to the NBP trend (between 24% and 31% of the trend due to all drivers). Changes in nitrogen addition (including fertilization and atmospheric deposition) is found to have only marginal net effect on NBP trends. However, this may not reflect reality because our model has only a very simple parameterization of nitrogen-effects on photosynthesis. The sum of NBP trends from each driver is larger than the trend obtained when all drivers are varied together, leaving a residual - non-attributed - term (22% - 26% of the trend due to all drivers) indicating negative interactions between drivers.

Introduction

Grassland is not a natural vegetation type in Europe. Europe's grasslands were created and managed to feed livestock for producing meat and dairy products (so-called pasture). These grasslands are mowed for forage production and grazed by ruminant animals, often within the same farm, although in regions of intensive livestock production, animals are also fed substantial additional amounts of crop feedstuff products. Meanwhile, nitrogen-rich mineral and organic fertilizers (manure/slurry) are commonly applied to European grasslands to sustain meat and dairy production. The frequency and intensity of these agricultural practices, combined with climate change, are expected to strongly impact the carbon (C) balance of grasslands in Europe (Soussana *et al.*, 2007).

The annual C balance of managed grassland ecosystems (also called net biome production, *NBP*; here using the definition proposed by Schulze & Heimann, 1998; Buchmann & Schulze, 1999; and Chapin *et al.*, 2006) must account for not only the fluxes of CO_2 exchanged with the atmosphere (net ecosystem production, *NEP*), but also the land-atmosphere CO_2 fluxes caused by lateral carbon import and export due to management and other processes (e.g., C export to rivers and groundwater). NEP is determined by the difference between net primary productivity (NPP) and ecosystem-level heterotrophic respiration (R_h). NPP, indicating the C incorporated into plant biomass, is known to be sensitive to climate (Melillo *et al.*, 1993), atmospheric CO_2 concentration (e.g., Ainsworth & Long, 2005) and nitrogen availability (Le Bauer & Treseder, 2008; Xia & Wan, 2008). R_h is also controlled by climate (Rustad *et al.*, 2001) as well as by organic C and nitrogen availability, and micro-environmental conditions (soil physical and chemical properties such as clay content, pH, etc), while organic C input to

R_h (including above and belowground litter) is determined both by NPP, and by C input (as manure) and exported as harvested biomass or ingested by grazing animals.

The amount of C exported from a grassland ecosystem depends on the grass consumed by ruminant livestock. According to the FAOstat agricultural statistics, during the period 1991-2010, a more than 17% reduction of livestock numbers occurred across Europe. This reduced the requirement for grass forage; at the same time grassland potential productivity was increasing (Chang *et al*., 2015a). These changes affected the ecosystem C balance, decreasing C export and increasing litter-fall to soils. Meanwhile, according to the harmonized high resolution land-cover change data set *HILDA* (Fuchs *et al*., 2013), the grassland area increased by 3.7% in the 30 European countries of the EU28 plus Norway and Switzerland. This figure includes concurrent loss and creation of grasslands by 5.8% and 9.5% of total grassland area in 1991, respectively, but with significant regional differences (EEA, 2005, pp. 47-53). The stronger relative reduction of livestock numbers compared to that of grassland area suggests an overall trend towards less intensive pasture usage.

A significant net C sequestration (a positive NBP of 15 ± 7 g C m^{-2} yr^{-1}) by grassland ecosystems of the 30 European countries during the period 1961-2010 was estimated in a previous study, using the process-based biogeochemical model ORCHIDEE-GM, which has an explicit and rather detailed representation of grassland management (Chang *et al*., 2015b). ORCHIDEE-GM simulates a significant *increase* of NBP with time during the past five decades (NBP linear trend of 0.25 ± 0.08 g C m^{-2} yr^{-2}, P = 0.26) with the rate of increase being much larger after 1990 (1.83 ± 0.30 g C m^{-2} yr^{-2}, P = 0.07) than before that date (-0.25 ± 0.15 g C m^{-2} yr^{-2}, P = 0.55). This acceleration of C sequestration can be attributed to changes in climate, CO_2 concentration, nitrogen atmospheric deposition, land-cover change, and to changes in management drivers, e.g., fertilization and decreasing grazing intensity, through decreased livestock numbers (Chang *et al*., 2015b). Yet, the quantitative contribution of these drivers and their interactions, to the observed NPP and NBP trends are not clearly understood.

In this study, we use the ORCHIDEE-GM Version 2.1 managed grassland model to analyze the overall trend of NBP of European grasslands during the past two decades. We separate the effects of changes in management, i.e., grassland management intensity (including abandonment, grazing intensity decrease or intensification, and change in fertilizer usage), and new grassland establishment (land-cover change) from the effects of observed external drivers, i.e., climate change, rising CO_2 and nitrogen deposition.

Material and methods

Model description

ORCHIDEE is a process-based ecosystem model for simulating carbon cycling in ecosystems, and water and energy fluxes from site-level to global scale (Krinner *et al*., 2005; Ciais *et al*., 2005; Piao *et al*., 2007). ORCHIDEE-GM (Chang *et al*., 2013) is a version of ORCHIDEE specifically developed to study grassland management. It incorporates a grassland module from the PaSim model (Reido *et al*., 1998; Vuichard *et al*., 2007a,b; Graux *et al*., 2011). ORCHIDEE-GM Version 1 was evaluated and some of its parameters calibrated using eddy covariance NEE and biomass measurements from 11 European grassland sites representative of a range of management practices. The model simulated the average NBP of these managed grasslands as 37 ± 30 g C m^{-2} yr^{-1}, P < 0.01; Chang *et al*., 2013). At continental scale, ORCHIDEE-GM Version 2.1 was first applied over Europe to calculate the spatial pattern, long-term evolution and interannual variability of potential productivity (potential productivity is the productivity resulting from the optimal management regime that maximizes livestock densities; Chang *et al*., 2015a). Chang *et al*. (2015a) further added a new parameterization to describe an adaptive management strategy whereby farmers react to a climate-driven change of previous-year productivity. Though a full nitrogen cycle is not included in ORCHIDEE-GM, the positive effect of nitrogen addition on grass photosynthesis, and thus on subsequent ecosystem carbon balances, is parameterized with a simple empirical function calibrated from literature estimates (Chang *et al*., 2015a). In a recent study, ORCHIDEE-GM Version 2.1 was used to simulate long-term NBP changes over European grasslands during the past five decades, indicating an enhancement of the C sink over the period 1991-2010 (with an NBP increasing rate of 1.83 ± 0.30 g C m^{-2} yr^{-2}, P = 0.07; Chang *et al*., 2015b).

Simulation set-up

Five drivers were considered for their impact on modeled NBP trend: 1) climate change, 2) rising global CO_2 concentration, 3) changes of nitrogen addition (including fertilization and atmospheric deposition), 4) land-cover change related to new grassland establishment and 5) the changes in grassland management intensity, through the observed reduction of ruminant livestock density (Chang *et al*., 2015b). These drivers were prescribed in ORCHIDEE-GM. Harmonized climate forcing data were taken from the ERA-WATCH reanalysis for the period 1901–2010, at a spatial resolution of 25 km × 25 km (Beer *et al*., 2014). Global atmospheric CO_2 concentration was from the combination of ice core records and atmospheric observations assembled by Keeling *et al*. (2009 and update). Yearly gridded mineral fertilizer and manure nitrogen application rates for grasslands in the European Union (EU27) were estimated by the CAPRI model (Leip *et al*., 2011, 2014) based on combined information from different data sources such as Eurostat, FAOstat and OECD, and spatially dis-aggregated using the methodology described by Leip *et al*. (2008). Gridded atmospheric nitrogen deposition rates for Europe from the European Monitoring & Evaluation Program (EMEP) data set were downloaded from the EU-PF7 GHG-Europe project (data available at http://gaia.agraria.unitus.it/ghg-europe/data/others-data) with the decadal means linearly interpolated to annual values.

Maps of changing grassland management intensity at 25 km resolution were constructed to drive the model with yearly changes in relative management intensity (fraction of extensively versus intensively managed grasslands) from 1961 to 2010, constrained by the total forage requirement of grass-fed livestock numbers (see Chang *et al*., 2015b for a detailed explanation of the calculation of the forage requirement and the diagnostic of the fraction of animals that receive complementary crop feed products). These maps were incorporated into the *HILDA* land-cover data set (Fuchs *et al*., 2013) to form enhanced historic land-cover maps delineating grassland management intensity and land-cover transitions (Chang *et al*., 2015b). Here if a fraction of grasslands in a 25 km grid cell is converted into another biome, it is no longer counted in the simulated NBP of grasslands, even though ORCHIDEE-GM simulates the C balance of the new biome. If a fraction of new grassland is created in a grid cell it is incorporated into the NBP by calculating at each time step the C balance of soils from the previous ecosystem to which new litter input from grassland is added.

The management intensity maps used above are constrained by the total forage requirement of grass-fed livestock numbers converted from the metabolizable energy (ME) requirement. The calculation of ME requirement depends on animal performance data such as liveweight and average daily milk production for each animal category (see Supporting Information Text S1 from Chang *et al*., 2015b), and a typical energy density of the feed suggested by IPCC (2006) is applied to convert energy requirement to forage requirement for the whole period 1961-2010. An assumption underlying the above calculations is that both ruminant diet composition (i.e., the fraction of arable crop-feed, crop by-products, and grass forage) and feed conversion efficiency (i.e., the animals' efficiency at converting feed mass into increases of the desired animal products such as meat and milk; referred to as FCE hereafter) are kept constant during the period 1961-2010. However, feedstuffs (including arable crop and crop by-products used to feed farm animals) have varied in the past, and FCE is changing due to the improved and balanced feeding practices and improved breeding which enable more of the feed to go to meat and milk production rather than to maintenance of the animals (Bouwman *et al*., 2005).

By applying a simple feed model (Ciais *et al*., 2007), grain-feed consumption per ruminant animal (including maize and other cereal grain used as feedstuff) is estimated to have been increasing rapidly over the past two decades (see Supporting Information Text S1 for detail), which indicates a possible decrease in the fraction of grass forage in ruminant diet (i.e., a decline in grass-based feed consumption per ruminant animal; here, grass-based feed includes fresh grass, hay and silage) during the same period. However, the fraction of grass-based feed in the ruminant diet cannot be calculated simply by subtracting feedstuff fractions in ruminant diet, because 1) the feedstuffs considered in this study are not complete; and 2) the simple feed model developed by Ciais *et al*. (2007) is purely diagnostic (based on animal and production data and a set of rules) thus it cannot produce by itself an increasing share of crop-based feed (unless it is in the input data). In addition, data on the past development of FCE do not exist for all European countries.

Given: 1) the increasing grain-feed consumption per ruminant livestock, and 2) the possible growth of FCE (Bouwman *et al*., 2005), we can make a new assumption to account for their impact on changes in total grass forage requirement and further on changes in grassland management intensity during the two most recent decades. The above two facts together are assumed to be fully responsible for the increase in meat and milk productivities of ruminant livestock after 1991, which implies the meat and milk productivities of ruminant livestock are kept constant in the calculation of the ME requirement (dashed lines in Fig. S1 as an example of beef cattle and cows). The new estimate of ME requirement is then used to calculate the observed grass-fed livestock numbers (dashed lines in Fig. S2), and further, to establish the new maps of changing grassland management intensity considering changes in crop-based feed per animal and FCE. The newly calculated observed grass-fed livestock numbers show a stronger decline (-25% in Livestock Units, LU) compared to the ones that do not consider the increase in crop-based feed consumption per animal and the growth in FCE (-18% in LU).

These new management intensity maps were also incorporated into the *HILDA* land-cover data set (Fuchs *et al*., 2013) to form a new version (Version 2) of the enhanced historic land-cover maps delineating grassland management intensity. The original maps with constant ruminant diet composition and FCE from Chang *et al*. (2015b) will be referred as Version 1 hereafter.

To assess the contribution of each of the five drivers (management intensity, land-cover change that forms new grasslands, climate change, rising CO_2 and changes in nitrogen addition) and possible interactions between them, we generated a series of ORCHIDEE-GM factorial simulations where one driver remains fixed while the others vary during the period 1991-2010. The simulation protocol is shown in Fig. 1. For the spin-up and the historic simulation (before 1991), the simulation was carried out exactly as detailed by Chang *et al*. (2015b) so that carbon stocks and fluxes in the starting year are already out of equilibrium, accounting for previous management, climate, nitrogen and CO_2 history. ORCHIDEE-GM was then run on each grid point at 25 km resolution during the period 1991-2010 forced by increasing CO_2, observed climate variability and nitrogen addition, with the adaptive management change algorithm described by Chang *et al*. (2015a), gridded land-cover change and annual changes in grassland management, but with constant ruminant diet composition and FCE (Version 1). The simulation with all drivers varying defines the control experiment (E_{CTL}). All the factorial sensitivity simulations are carried out for the period 1991-2010, starting from the same state (year 1991) as in E_{CTL} (e.g., soil and vegetation carbon pools, optimal animal stocking rates (S_{opt}), and management intensity maps). In each factorial simulation (*E1* to *E5*; Fig. 1), one of the five drivers is held constant at its value in year 1991, while all other drivers vary as in the control run. In the case of the "constant climate" driver (*E1*), in order to keep interannual variability, we cycled the climate fields from years 1991 to 1995 in a loop. Another set of simulations (Set-2, indicated with a prime, viz: E_{CTL}' and $E1'$ to

E5′) were conducted with the same protocol described above, but using the grassland management intensity maps Version 2. Differences between the two sets of simulations allow us to assess the contribution of each of the five drivers over the full range of uncertainty due to the time-variable ruminant diet composition and FCE. All results from simulation Set-2 (i.e., using Version 2 of the historic management and land-cover change maps; E_{CTL}' and *E1′* to *E5′*) will be marked "in Set-2".

Trends in the drivers during past two decades

During the period 1991-2010, mean annual temperature increased in Europe. The fastest rates were found in southern Spain and southeastern Europe (Bulgaria and Romania), where the warming rate was 0.7 ^{o}C per decade (Fig. S3a). Concurrently, total annual precipitation increased in many regions of Europe, but a decline in precipitation occurred in the west of Ireland, north of Spain, southern France, and the west of Italy and Austria (Fig. S3b).

Large amount of nitrogen fertilizer (more than 60 kg N ha^{-1} yr^{-1} typically) have been applied to grassland in Germany, the Netherlands, Belgium, Luxembourg, France and Ireland, whereas application rates have remained low in other regions (mostly less than 40 kg N ha^{-1} yr^{-1}; see Fig. S1 of Chang *et al*., 2015a). A decrease in nitrogen fertilization during the period 1991-2000 is present in the gridded nitrogen addition maps used as input to ORCHIDEE-GM (Fig. S3c). According to the European Monitoring & Evaluation Programme (EMEP) data set cited above, atmospheric nitrogen deposition rates for Europe increased over part of western Europe during the period 1991-2000, but decreased over eastern Europe for the same period (Fig. S3d).

According to the *HILDA* land-cover data set, the area of grassland remained fairly stable during the period 1991-2000 in northern and western Europe, but increased in Portugal, Spain, Italy and eastern Europe countries; it decreased in Slovenia (Fig. S3e). In parallel, a decline in the area of grassland in the Czech Republic is shown in this satellite-derived data set, but this may not be real, because an extension in pasture during the period 1990-2000 may be expected due to government policies to keep farmland managed as pasture wherever possible (EEA, 2005). Ruminant livestock numbers have been reduced by 17% during the period 1991-2010 in Europe (EU28 plus Norway and Switzerland; FAOstat). Large reductions took place in eastern European countries (Fig. S3f) in response to the major political changes which happened in the early 1990s. The reduction in ruminant livestock numbers as well as the change in grassland area during the past two decades has mainly driven a transition from intensively managed grasslands to more extensively managed ones (Fig. S3g). This transition is even more severe (Fig. S3h) when the lower grass forage requirement is considered (i.e., less grass-fed livestock numbers; dashed line in Fig. S2), i.e., the possible decrease of grass-feed fraction in ruminant diet (given the increasing grain-feed consumption per ruminant animal; Fig. S4c) and the growth in FCE (Bouwman *et al*., 2005).

Carbon balance of European grasslands

The net ecosystem production (NEP) is defined hereafter as the difference between gross primary production (GPP) and ecosystem respiration (R_{eco}) as suggested by Chapin *et al*. (2006):

$$NEP = GPP - R_{eco} \tag{1}$$

where R_{eco} is the respiration of all organisms including autotrophic respiration (R_a) by primary producers and heterotrophic respiration (R_h) by heterotrophic organisms (microbes in animals' digestive systems, and microbes in soils). Since net primary production (NPP) is defined as GPP minus R_a, NEP thus can also be expressed as:

$$NEP = NPP - R_h \quad (2)$$

R_h simulated by ORCHIDEE-GM is defined by heterotrophic respiration caused by grazing animals (microbes living in animals' digestive systems) and by soil microbes. A positive NEP value indicates a net removal (sink) of atmospheric CO_2. Note that in ORCHIDEE-GM there is no explicit representation of microbial biomass but rather the model uses first order kinetics to represent the decomposition of organic matter (Krinner *et al.*, 2005).

NBP denotes the total rate of C accumulation (or loss) from ecosystems at large spatial scales (e.g., a grid cell of 25 km or a region), and is defined by:

$$NBP = NEP + C_{input} - C_{export} \quad (3)$$

where C_{input} is the flux of C entering the grassland ecosystem through manure and/or slurry application, and C_{export} is the total C lost from the grassland ecosystem through plant biomass export (mowing), and CH_4 emissions by grazing animals. C_{export} through milk production and animal body-mass increase is not determined and will be neglected for the calculation of NBP, which has only marginal effect on the NBP estimate (Chang *et al.*, 2015b). It must be noted that C lost through dissolved organic (DOC) and carbonate-borne inorganic (carbonate-borne DIC) leaching to rivers is not determined and will also be neglected for this study in the estimation of NBP, whereas biogenic DIC (formed by CO_2 from soil respiration, including R_a by root and R_h by heterotrophic organisms; Kindler *et al.*, 2011) has been implicitly accounted for in the simulated R_a and R_h fluxes to the atmosphere.

Attribution method of NBP trends

The net effect of a factor or driver *x*, is defined as the difference of NBP trend between the control simulation and each factorial simulation, Δ_x, as:

$$\Delta_x = N\dot{B}P_{CTL} - N\dot{B}P_x \qquad (4)$$

where $N\dot{B}P_{CTL}$ is the NBP linear trend during 1991-2010 from the control simulation where all drivers including *x* are varied, and $N\dot{B}P_x$ is the NBP linear trend of simulation *x* where only this driver remains fixed. Therefore, $\Delta_{climate}$, Δ_{CO2}, $\Delta_{nitrogen}$, Δ_{LCC} and $\Delta_{management}$ are here the individual effects of changes in climate, CO_2, nitrogen fertilization, grassland area (new grasslands produced from land-cover change; *LCC*) and in management intensity on the NBP linear trend. The sum of individual effects can be less than or more than the effect of all the factors taken together, due to non-linear interactions, and a residual, non-attributed, is defined as $\Delta_{residual}$. The same attribution method was also used to determine the effect of each factor on the trends of NEP (which can be divided into NPP and R_h), C_{input} and C_{export}.

Results

Trends in NBP and in its component

During the past two decades, NBP of European grasslands simulated by ORCHIDEE-GM has increased at mean rate of 1.8 to 2.0 (in Set-2) g C m^{-2} yr^{-2} (Table 1). Given a decline of carbon input from organic fertilizers (C_{input}; Table 1 and Fig. 2b), this increasing NBP must come from an enhanced CO_2 fixation (NEP) and/or the reduction of carbon exported as forage and CH_4 emissions by grazing animals (C_{export}; Table 1 and Fig. 2b). The trend in NEP is derived from the trend of NPP minus the one in R_h (Table 1 and Fig. 2a). A more intense increase in NBP (2.0 g C m^{-2} yer^{-2}) found in simulation Set-2 mainly resulted from the faster decrease of C_{export} (with a rate of -1.2 g C m^{-2} yr^{-2}; Table 1) caused by the stronger decrease in total forage requirement when accounting for the possible decrease of grass-feed fraction in ruminant diet and the growth in FCE.

Attribution of NBP trends to different drivers

All drivers considered in this study have the overall effect of increasing NBP (positive trend) for the whole European grassland ecosystem, during the period 1991-2010 (Table 1; Fig. 3). The key result is that changes in grassland management intensity make the largest contribution to the average NBP trend, namely: 0.65 - 0.88 (in Set-2) g C m^{-2} yr^{-2}, about 36% - 43% (in Set-2) of the trend due to all drivers. Management intensity changes alone are estimated to have enhanced soil carbon stocks by 164 - 214 (in Set-2) Tg C in 20 years over 1.3×10^6 km^2 grassland (i.e., 0.23 - 0.30 (in Set-2) Mg CO_2 ha^{-1} yr^{-1} when converted to the C sequestration per hectare of grassland). Newly established grassland (increase of grassland area) caused a comparable but lower NBP positive trend of 0.48 (in Set-2) - 0.51 g C m^{-2} yr^{-2} (24% (in Set-2) - 28% of the trend due to all drivers). The increase in NBP attributed to increasing CO_2 and climate change is 0.54 (in Set-2) - 0.55 g C m^{-2} yr^{-2} (27% (in Set-2) - 30% of the trend due to all drivers) and 0.54 (in Set-2) - 0.56 g C m^{-2} yr^{-2} (27% (in Set-2) - 31% of the trend due to all drivers) respectively. The net effect of changes of nitrogen addition is only marginal (2% of the trend due to all drivers). The sum of NBP trends attributed to each driver is larger than the overall NBP trend caused by all drivers, leaving a residual term (about 22% (in Set-2) - 26% of the trend due to all drivers). This residual is due to negative interactions between the effects of the different drivers.

Trends of grassland management intensity are also the largest factor explaining C_{export} trends (Table 1). Less intensively managed grassland (Figs S3g and S3h) tends to reduce C_{export} because average stocking rates decrease, which enhances NBP by higher litter-fall and soil carbon storage. Climate change and rising CO_2 concentration both have a positive effect on NEP trends, by enhancing NPP in excess of R_h, but they also tend to increase C_{export} due to a higher NPP being available for forage. Grassland establishment (land-cover change from cropland/forest to grassland) has a moderate and positive effect on NEP, however, because R_h decreases more than NPP (Table 1). This lower R_h in newly established grassland is due to the usually lower soil carbon storage of the source land cover, which is usually cropland (Post & Kwon, 2000; Guo & Gifford, 2002). This result depends on the source ecosystem and on the simulation by ORCHIDEE of its soil C stock. In ORCHIDEE, croplands are harvested so that only a small fraction of biomass returns to the soil; this tends to make soil C stocks lower than those in grasslands.

Spatial distribution of NBP trend and its attribution

Figure 4 shows the spatial distribution of the simulated trends in NBP across European grasslands during 1991-2010 from simulation Set-1 (the spatial pattern from simulation Set-2 is very similar to that of simulation Set-1). The simulated trend of NBP is positive over 80% of European grasslands. After aggregation into the major agricultural regions defined in Table S1 (for details see Olesen & Bindi, 2002), the largest positive NBP trends appear over northeastern, and eastern (Figs 3 and 4) Europe, where decreasing management intensity and grassland establishment are the major causes of positive NBP trends. Positive NBP trends larger than 2.0 g C m^{-2} yr^{-2} were also found in the British Isles and southeastern Europe, where changes in grassland management intensity play the most important role in explaining positive NBP trends. In the British Isles climate change has the largest positive effect compared to other regions of Europe (Figs 3 and 5a). It is noteworthy that in the British Isles, the sum of the five effects is 49% (in Set-2) - 50% larger than the NBP trend obtained in the control simulation ($N\dot{B}P_{CTL}$), suggesting a strong negative effect due to their interactions (Fig. 3). Not all the regions show an increasing NBP, and we obtained a decline of NBP (negative trend) in northwest Germany, France, Spain and southwest Romania (Fig. 4). A moderate NBP increase was simulated in alpine regions attributed to the effect of rising CO_2 and changes in management intensity.

Within the five drivers considered in this study, changes in management intensity (mainly changing from intensively managed grasslands to extensively managed ones; Chang *et al.*, 2015b) is clearly a major factor causing NBP to increase in those regions where it did (e.g., the British Isles, alpine regions, and eastern EU countries; Figs 3 and 5d). It caused simultaneously a decline in NEP which forced NBP to decrease (Fig. 6b), but caused an even stronger reduction of C_{export}, which compensates for decreased NEP and results in a net positive trend in NBP (Fig. 6d). Grassland establishment diagnosed from high resolution land-cover change satellite data has a positive effect on NBP trends over regions where an increase of grassland area happened (Portugal, northeastern, southeastern and eastern European countries, except the Czech Republic; Fig. S3e), but also can have a negative effect on NBP trends, as in Finland (Fig. 5c), because the soil C stock of the source ecosystem (forest in Finland) is larger than that of grassland and during 2001-2010 the newly established grassland in Finland was mainly converted from forest. Regionally, climate change can have either a positive or a negative effect on NBP trends (Fig. 5a), mainly through its effect on NEP. This effect can be seen from the similar spatial patterns of NEP trends and NBP trends due to climate change (Fig. 6a). The difference between NBP and NEP trend patterns are located in the most productive grasslands such as in Denmark, the Netherlands, and Belgium, where enhanced C_{export} offsets or even exceeds the effects of enhanced NEP, thus causing a neutral, or negative, effect of climate change on NBP in those regions. Rising CO_2 concentration has a moderate and positive effect all over Europe, whereas changes in nitrogen fertilization only have a marginal effect on NBP trends (Fig. 5b).

Discussion

In this study, we attributed the NBP trend over European grasslands to its driving factors through a series of simulations of the ORCHIDEE-GM model separately quantifying the effect of each factor. In summary, four of the five factors considered in this study (except nitrogen addition) resulted into an overall positive effect on the NBP trend, but with different spatial patterns, magnitudes and mechanisms.

Climate change during the past two decades was simulated to cause an increase in NBP over European grassland as a whole, by increasing NPP more than R_h (Table 1). However, the effect of recent climate trends can be positive or negative in different regions (Fig. 6a). For example, climate change induced a positive trend of NBP in the Nordic countries and the British Isles, likely from temperature warming, which enhances plant growth and extends the growing-season length in cold environments (Fig. S5). Our modeled positive effect of climate change on productivity (NPP) in the British Isles is consistent with the increasing aboveground live biomass from 1982 to 2006 derived from remote sensing products, which shows positive correlation with annual mean temperature (Xia *et al*., 2014). In regions where productivity is limited by water stress (Le Houerou *et al*., 1988; Knapp *et al*., 2001; Nippert *et al*., 2006), a decline in precipitation by more than 75 mm yr^{-1} per decade (such as in north Spain, southern France and northeastern Italy, Fig. S3b) or a strong warming by more than 0.7 ^{o}C per decade (such as in Romania and north Bulgaria) cause drought conditions that reduce productivity and furthermore reduce NBP (Fig. 6a).

Elevated CO_2 concentration has the dual effect of increasing leaf photosynthesis and reducing stomatal conductance, thus indirectly increasing soil moisture in unsaturated soils. These effects increase water-use efficiency (Rötter & van de Geijn, 1999) and reduce the consumption of soil moisture by plant transpiration (Soussana & Luescher, 2007). In our simulation, rising CO_2 concentration causes a 4.5% increase in grassland water-use efficiency (defined as the ratio of GPP to transpiration) during the two most recent decades. This increase is calculated from the simulation E1 in which the climate of 1991-1995 is recycled, but CO_2 concentration rises (Fig. 1). Increases in productivity of temperate grassland stimulated by rising CO_2 concentration has been observed in FACE experiments (e.g., BioCON and Swiss FACE; Ainsworth *et al*., 2003; Ainsworth & Long, 2005) and also been simulated by ORCHIDEE-GM (Chang *et al*., 2015a). In a separate test, ORCHIDEE-GM simulated an increase in aboveground dry matter production of C3 grass of 10.7% under elevated CO_2 of 550 ppm (Chang *et al*., 2015), which is close to that from FACE experiments, e.g., the increase of 10.5% observed by Ainsworth & Long (2005). However, trends and variability in temperature and precipitation, as well as nitrogen limitation, will all interact with the effects of elevated CO_2 in the future to determine actual changes in grassland productivity in response to CO_2 (Jones & Donnelly, 2004; Soussana & Luescher, 2007). The residual effect on NBP trends found in this study (Table 1, Fig. 3) may be partly attributed to these interactions.

Application of nitrogen to grassland improves soil nitrogen availability, leading to higher nitrogen concentration in plant leaves, thus enhancing plant growth and productivity (Frink *et al*., 1999). ORCHIDEE-GM accounts for the positive effect of nitrogen addition on photosynthesis (Chang *et al*., 2015a). However, the effect of nitrogen addition remains marginal, because: 1) the rates of change in nitrogen addition (including fertilization and atmospheric deposition) are small over most European grasslands (within ±6% per decade, Fig. S5b); or 2) nitrogen addition rate is already high throughout the period (e.g., over 100 kg N ha^{-1} yr^{-1} in the south of Ireland, France, the Netherlands and eastern Germany; Fig. S6a) in the regions with higher rates of change (more than 6% per decade, Fig. S6b). As a result, the net effect of changes in nitrogen addition (including fertilization and atmospheric deposition) on NBP trends is found to be only marginal. However, this finding may not reflect reality, since our model has only a very simple parameterization of the effects of nitrogen on photosynthesis (Chang *et al*., 2015a). In fact, omitting nitrogen addition from our model could also have effect on allocation (Poorter *et al*., 2012), and possibly affect soil organic decomposition and hence heterotrophic respiration (see review by Janssens *et al*., 2010 for forest).

Note that in our simulations, climate change, rising CO_2 concentration and changes in nitrogen addition all impact NBP trends only through their indirect effect on NPP. The NPP change (increase/decrease) will cause a trend of fresh organic C availability to the three soil carbon pools of ORCHIDEE-GM, thus affecting R_h. In our model, R_h depends on the soil organic C formed during the historical period (pre-1991), and on soil temperature and moisture trends. Overall R_h tracks the NPP change, but with a lag. As a result, the nonsynchronous evolution of NPP and R_h causes the NEP trend (and further the NBP trend). However, several other processes that could affect R_h are omitted from our model: 1) the dependence of R_h not only on soil organic C availability, but also on microbes which are assumed to be the direct producer of R_h and whose activity can be temperature dependent (Allison *et al*., 2010); 2) the priming effect that emphasizes that increased inputs of fresh C (e.g., increased NPP due to climate change, rising CO_2 and nitrogen addition) could stimulate soil microbes to decompose old soil organic matter (Kuzyakov *et al*., 2000; Kuzyakov, 2010); although an experiment on grassland soil in central France suggested high nutrient availability might reduce the priming effect, thus increase the mean residence time of soil C (Fontaine *et al*., 2011); 3) elevated CO_2 might affect soil microbial community structure (e.g., Janus *et al*., 2005; Carney *et al*., 2007; Guenet *et al*., 2012) and possibly enzyme activities (though conflicting effects have been reported; Freeman *et al*., 2004), might further impact soil organic C decomposition. The soil microbial community structure could alter the temperature sensitivity of R_h (Bradford *et al*., 2008), and the enzyme activities that are directly responsible for R_h. We need better models of soil C cycling and the representation of these microbial mechanisms should be a high priority in future model development. The CLM microbial model (Wieder *et al*., 2013) shows one way forward.

To avoid the negative side-effects of some farming practices, since 1962 the European Union (EU) has provided various incentives to farmers through the Common Agricultural Policy (European CAP). In 1984 the European Community introduced milk production quotas that contributed to a reduction in the dairy cow population in Europe. This was followed in 1991 by the Nitrates Directive (91/676/EEC) that restricted the application of animal manure in "nitrate vulnerable zones" (46.7% of EU-27 land area in 2012; European Commission, 2013) to a maximum of 170 kg N ha^{-1} yr^{-1}, effectively capping livestock density in pastures at some 1.7 livestock units (LU) per hectare (Annex 1 in Webb *et al*., 2011). In 1992, the incentives of CAP shifted from price support to direct aid payments to farmers who withdraw land from production and further limit stocking levels. As a result of these policies, the livestock numbers in Europe have decreased. In addition, major political changes in eastern and central Europe also resulted in wet grasslands being abandoned (Joyce, 2014). The European-wide livestock numbers declined by more than 17% during the period 1991-2010 (FAOstat), reducing the requirement for grass forage and for the grassland C balance, less forage means less C export and thus increasing NBP. Our simulation is forced by the observed decrease of grass-fed livestock numbers (decline by 18% - 25% (in Set-2); Figs S1 and S2) in each European region, and takes into account the NBP response to a less intensive grassland management through its constraint that the total forage requirement of grass-fed livestock must be satisfied by grass NPP (cut and grazed). Without harvest by mowing or grazing, grasslands with less animals can have higher leaf area index (LAI), thus higher NPP than more intensively managed ones (Joyce, 2014; also see Chang *et al*., 2015b). In this study, we simulated an annual mean LAI of extensively managed grassland 26% higher than that of intensively managed grassland over Europe (data averaged for the period 1991-2010). The extra C taken up would be accumulated in soil as litter instead of being exported as forage. Litter has a relatively longer turnover time than forage (most forage C is consumed then returned to atmosphere within one year). As a result, the changes in grassland management in Europe, characterized by changing from intensively to extensively managed grassland (Fig. S2; Figs S3f, g and h) are able to cause enhanced sequestration of C in soil (NBP increase). Furthermore, during the transition from intensively to extensively managed grassland, nitrogen accumulated in managed grassland soils (e.g., due to fertilization) may maintain a high productivity (i.e., NPP) for some years. This residual effect of nitrogen fertilization on productivity was not taken into account because the nitrogen-effects on photosynthesis in ORCHIDEE-GM (Chang *et al*., 2015a) will immediately stop when grassland is converted to extensive management. In this case, our model may underestimate the NPP increase caused by decline in management intensity, and further underestimate the positive trend of NEP and NBP in the control simulation ($N\dot{B}P_{CTL}$). It implies the effect of changes in management intensity in reality could be even larger than that estimated here (given by Eqn 4: $N\dot{B}P_{CTL}$ - $N\dot{B}P_{management}$; about 36% - 43% (in Set-2) of the trend due to all drivers). Thus again, fully accounting for the nitrogen cycle is required to produce better estimates of the grassland C balance and its trend. Nevertheless, a large portion of the NBP trend (36% - 43% of the trend due to all drivers) can be attributed to the reduction of grassland management intensity in Europe, that is probably caused by changes in policy and socio-economic influences. As fas as we are aware this is the first instance of a modeling study revealing the impact of policy on the C balance of European grasslands.

ORCHIDEE-GM estimates an average NBP of 19 to 21 (in Set-2) ± 7 gC m^{-2} yr^{-1} in the most recent two decades -- a cumulative C sink of 1.8 to 1.9 (in Set-2) ± 0.7 gigatonnes (Gt) CO_2 over about 1.3×10^6 km^2 for European grasslands during 1991-2010 (EU28 plus Norway and Switzerland). This amount offsets about 10% of total greenhouse gases (GHGs) emissions in the whole agricultural sector during this period (a total of 17.3 Gt CO_2 equivalent from 1991-2010 including CO_2, CH_4 and N_2O emissions; data from FAOstat). With respect to European grassland ecosystems, changes in grassland management alone were estimated to have enhanced the soil C sequestration during the 20-year period studied by 0.60 - 0.79 (in Set-2) Gt CO_2 and to have simultaneously reduced CH_4 and N_2O emissions by 0.42 – 0.48 (in Set-2) Gt CO_2-equivalent and by 0.22 - 0.27 (in Set-2) Gt CO_2-equivalent respectively, offsetting 7% - 9% (in Set-2) of GHG emissions in the whole agricultural sector during this period, making a substantial contribution to climate change mitigation (Ripple *et al.*, 2014).

The C sequestration enhanced by changes in grassland management alone (0.23 - 0.30 Mg CO_2 ha^{-1} yr^{-1}) is close to the C sequestration potential from optimizing grazing management for rangeland estimated by Henderson *et al.* (2015), which is 0.17 – 0.32 Mg CO_2 ha^{-1} yr^{-1} (to be consistent in regions considered by Henderson *et al.* (2015), values in western Europe and in eastern Europe and Russia were used), but higher than that for pasture in the same estimation (0.03 – 0.05 Mg CO_2 ha^{-1} yr^{-1}). It is noteworthy that the area of amenable grassland in Henderson *et al.*, 2015 is only 13% of grassland area in our simulation. Our estimation is also within the range of the global mean C sequestration potential due to grazing management (per hectare bases) reported by Lal (2004; 0.18 - 0.55 Mg CO_2 ha^{-1} yr^{-1}) and Smith *et al.* (2008; 0.11-0.81 Mg CO_2 ha^{-1} yr^{-1}), while the value from Smith *et al.* (2008) considered a broader range of practices (grazing, fertilization and fire). Furthermore, the differences between our estimate and that from literature above should be kept in mind: 1) in this study, the C sequestration enhanced by changes in management intensity accounts for climate change, rising CO_2 and a simple management change constrained simply by historic grass forage requirement; 2) Henderson *et al*. (2015) estimated C sequestration potential that could be achieved by optimizing grazing management based on assumed grazing management scenarios, and considered climate change at 0.5^o resolution; 3) the C sequestration potential estimated by Lal (2004) and Smith *et al*. (2008) are data-based (e.g., based on a small number of field studies), but didn't include the climate and CO_2 perturbation (i.e., they did not use gridded simulations accounting for climate gradients across a domain). Thus our estimate and the previous estimates above could be quite complementary allowing a better view to be developed of the role of grassland management on the C balance and further on GHG mitigation.

Soil carbon stocks increased because of decreased management intensity in our control simulation (originating from the combination of reduced ruminant livestock numbers and the changes in grassland area, where only the new grasslands creation were considered; Fig. S3e to h). Carbon sequestration efficiency due to changes of ruminant livestock density can thus be calculated as the ratio between NBP trend due to changes in management ($\mathit{\Delta}_{management}$, see '*Attribution method of NBP trends*' for detail) and trend of livestock density (Table 2). Here, $\mathit{\Delta}_{management}$ is the individual effect of changes in grassland management intensity on the NBP linear trend; livestock density over grassland is calculated as the ratio of grass-fed livestock numbers to grassland area; and carbon sequestration efficiency due to changes of ruminant livestock density measures how much extra C can be sequestrated in grassland soil per each LU reduction of livestock numbers. Averaged over European grasslands, a unit LU reduction of livestock numbers is able to enhance soil C sequestration by 1016 (in Set-2) - 1131 kg C per year (i.e., with carbon sequestration efficiency of 1016 (in Set-2) - 1131 kg C LU^{-1} yr^{-1}). This reduction indicates a substantial contribution to GHG mitigation (Ripple *et al*., 2014). High sequestration efficiency was found in the Nordic countries, the British Isles, alpine regions, southeastern and eastern regions, suggesting grassland ecosystems in these regions can benefit more, with regards to GHG mitigation, from each unit of livestock numbers reduction. However, the sequestration efficiency in the Nordic countries, the British Isles, and alpine regions from the simulation Set-1 are not given, and should be neglected because the too-low trends in livestock density (e.g., lower than -0.2 LU km^{-2} yr^{-1}) might cause unrealistically high sequestration efficiency. Grassland ecosystems in western, Mediterranean, and northeastern regions of Europe can also benefit from livestock number reduction of moderate magnitude.

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Supporting Information Legends

Text S1. The changes in feedstuffs for ruminants.

Table S1. Major agricultural regions in Europe (Olesen & Bindi, 2002).

Figure S1. The evolution of meat productivity of beef cattle and milk productivity of cows in Europe. Data are averaged for EU28 plus Norway and Switzerland. Solid lines indicate the productivities derived from FAOstat; dashed lines are the constant productivities of ruminant livestock in the new calculation of ME requirement assuming that the growth in feed conversion efficiency is consistent with the increase of meat and milk productivities of ruminant livestock after 1991.

Figure S2. Grass-fed livestock numbers in each of major agricultural regions and their evolution during the period 1961-2010. The numbers were converted to livestock unit (LU) based on the calculation of metabolizable energy (ME) requirement of each type of animal with variable (i.e., the growth in feed conversion efficiency is consistent to the increase of meat and milk productivities of ruminant livestock after 1991; dashed lines) or constant (solid lines) feed conversion efficiency.

Figure S3. Spatial distribution of the changing rate (linear trends) during the period 1991-2010 in: (a) mean annual temperature, (b) total annual precipitation, (c) nitrogen fertilization, (d) atmospheric nitrogen deposition, (e) grassland area, (f) ruminant livestock numbers and (g) in fraction of intensively managed grassland in total grassland area with constant feed conversion efficiency (FCE) or (h) with assumed changes in FCE. The changing rate in (c) nitrogen fertilization and (d), atmospheric nitrogen deposition are estimated for the period 1991-2000, because in the database they are assumed to be constant from 2000 till 2010. Temperature and precipitation were from ERA-WATCH reanalysis climate forcing data at a spatial resolution of 25 km (Beer *et al.*, 2014). Gridded mineral fertilizer and manure nitrogen application rate was estimated by the CAPRI model (Leip *et al.*, 2011, 2014), based on combined information from official and harmonized data sources such as Eurostat, FAOstat and OECD, and spatially dis-aggregated using the methodology described by Leip *et al.* (2008); grassland area was extracted from the HILDA data set (Fuchs *et al.*, 2013); ruminant livestock numbers were taken from FAOstat with annual country-averaged statistical data on major ruminant livestock numbers for dairy cows, beef cattle, sheep and goats; livestock species are converted to livestock unit (LU) based on the calculation of metabolizable energy requirement (see Supplementary Information Text S1 of Chang *et al.*, 2015); fraction of intensively managed grassland in total grassland area is as established by Chang *et al.* (2015), constrained by the total forage requirement (derived from metabolizable energy requirement) of grass-fed livestock numbers.

Figure S4. Temporal evolution of (a) total feedstuff products, (b) farm animal numbers, and (c) feedstuff for ruminant and grain-feed consumption per head of ruminant during the last five decades. For the feedstuffs, cereal grains include maize and other cereals; other crop products and by-products included cakes of cereals and oilseeds, brans, and pulses; grain for cattle was the residual grain-feed for cattle after being distributed successively to poultry and pigs using the simple feed model (Ciais *et al.*, 2007). To keep data consistency, the figure shows the total quantities from 23 countries of Europe, where data from Croatia, Czech Republic, Estonia, Latvia, Lithuania, Slovakia, and Slovenia were not included in due to the short period of data availability.

Figure S5. Shift in seasonal evolution of grassland GPP during the last two decades in the Nordic countries and the British Isles. The monthly mean GPP of grassland was simulated by ORCHIDEE-GM, aggregated and averaged over each region according to the area and the management intensity (extensively or intensively managed) of grassland in the enhanced historic land-cover maps delineating grassland management intensity (Version 1; see main text section '*Simulation set-up*' for detail). Decadal averages of monthly GPP (for the period 1991-2000 and 2001-2010 respectively) were used. GPP: gross primary production.

Figure S6. (a) mean nitrogen addition rate over European grassland (including fertilization and atmospheric deposition) and (b) its normalized changing rate during the period 1991-2000. The normalized changing rate of nitrogen addition is calculated as the ratio of changing rate (linear trend) to mean nitrogen addition rate.

Tables

Table 1. Trends in NBP and its components over European grasslands during the period 1991-2010, and the effects of the drivers on these trends.

	Linear trends	Effect of the drivers on the trends					
	(g C $m^{-2} yr^{-2}$)	Climate forcing	Atmospheric CO_2 concentration	Nitrogen addition	Grassland area	Grassland management	Residual
NBP	1.8 / 2.0'	0.6 / 0.5'	0.5	0.0	0.5	0.6 / 0.9'	0.5 / 0.4'
NEP	2.0 / 1.9'	0.9	0.8 / 0.7'	0.0	0.5	-0.1	0.1 / 0.0'
C_{export}	-0.1 / -0.4'	0.3	0.2	0.0	-0.1	-0.9 / -1.2'	-0.3 / -0.4'
C_{input}	-0.2 / -0.3'	0.0	0.0	0.0	0.0	-0.2	0.0
NPP	4.7 / 5.0'	2.7	2.0 / 1.9'	0.0	-0.2	0.7 / 1.0'	0.4
R_h	2.7 / 3.1'	1.8	1.2	0.0	-0.7	0.8 / 1.1'	0.3 / 0.4'

Note: Values without a prime (') are the results from the simulation Set-1 accounting for the constant ruminant diet composition and feed conversion efficiency; Values with a prime (') are the results from the simulation Set-2 accounting for the varied ruminant diet composition and feed conversion efficiency (section '*Simulation set-up*'); when results from the two sets of simulations are the same, only one value will be shown. NEP, net ecosystem production; C_{export}, carbon exported from grassland ecosystem as forage and CH_4 emission; C_{input}, carbon input by organic fertilizer application; NPP, net primary production; R_h, heterotrophic respiration. NEP is defined as the difference between NPP and R_h, indicating the CO_2 sequestration from atmosphere.

Table 2. Carbon sequestration efficiency due to the changes in livestock density over European grasslands.

Regions	$\Delta_{management}$ g C m^{-2} yr^{-2}	Trends in livestock density LU km^{-2} yr^{-1}	C sequestration efficiency kg C LU^{-1} yr^{-1}
Nordic	0.14 / 0.19'	-0.01 / -0.21'	-* / 1612'
British Isles	1.24 / 1.75'	-0.17 / -0.87'	-* / 2013
Western	0.36 / 0.55'	-0.46 / -0.75'	795 / 734'
Mediterranean	0.25 / 0.41'	-0.42 / -0.51'	605 / 795'
Alpine	0.48 / 0.75'	-0.16 / -0.49'	-* / 1539'
North eastern	0.90 / 1.53'	-1.07 / -1.77'	839 / 860'
Sourth eastern	1.82 / 1.95'	-0.96 / -1.08'	1884 / 1810'
Eastern	2.22 / 2.82'	-1.58 / -2.12'	1398 / 1330'
Total	0.65 / 0.88'	-0.58 / -0.87'	1131 / 1016'

* Carbon sequestration efficiency is not given due to the too-low trends in both NBP and/or livestock density.

Note: Values in the $\Delta_{management}$ column are the individual effects of changes in grassland management intensity on the NBP linear trend; Livestock density over grassland is calculated as the ratio of grass-fed livestock numbers to grassland area. Values without the prime (′) indicate the results from the simulation Set-1 accounting for the constant ruminant diet composition and feed conversion efficiency; Values with the prime (′) indicate the results from the simulation Set-2 accounting for the varied ruminant diet composition and feed conversion efficiency (section '*Simulation set-up*' and Supporting information Text S1 for detail).

Figure legends

Figure 1. Illustration of the simulation protocol and the five factors used as input data for various simulations. E_{CTL}: control simulation with all factors varied; E1–E5: the factorial sensitivity simulations started from the same state (at the end of 1990) as in E_{CTR}, but with one of the five drivers being held constant to the value of year 1991 or cycled in a loop with the climate fields from years 1991 to 1995 (in gray background). The NBP trend during 1991-2010 from simulation E_{CTL}, and *E1–E5* is expressed as $N\dot{B}P_{CTL}$, $N\dot{B}P_{climate}$, $N\dot{B}P_{CO2}$, $N\dot{B}P_{nitrogen}$, $N\dot{B}P_{LCC}$, and $N\dot{B}P_{management}$ respectively.

Figure 2. Changes in the components of NBP ((a) NPP and R_h, (b) C_{export} and C_{input}) across European grasslands during the period 1991-2010. NPP: net primary production; R_h: heterotrophic respiration; C_{export}: carbon exported from grassland ecosystem as forage; C_{input}: carbon input by organic fertilizer application. Results are derived from simulation Set-1 considering constant ruminant diet composition and feed conversion efficiency.

Figure 3. The NBP trends of grassland ecosystems and the effect of each driver considered in this study. The figure shows the results for all the grassland in Europe and for grassland in each major agricultural region (region 1 to 8 as shown in the figure; also see Table S1 for detail). $N\dot{B}P_{CTL}$ is the NBP trend during 1991-2010 from the control simulation; $\Delta_{climate}$, Δ_{CO2}, $\Delta_{nitrogen}$, Δ_{LCC} and $\Delta_{management}$ are the individual effects of climate change, rising CO_2 concentration, changes in nitrogen addition, in land cover (grassland area) and in grassland management intensity respectively to the NBP trend. The sum of individual effects can be less than, or more than, the effect of all the factors taken together, due to non-linear interactions, and the residual is defined as $\Delta_{residual}$. Bars filled with solid color indicate the trends and the effects from the simulation Set-1 accounting for the constant ruminant diet composition and feed conversion efficiency; bars filled with parallel lines indicate the trends and the effects from the simulation Set-2 accounting for the varied ruminant diet composition and feed

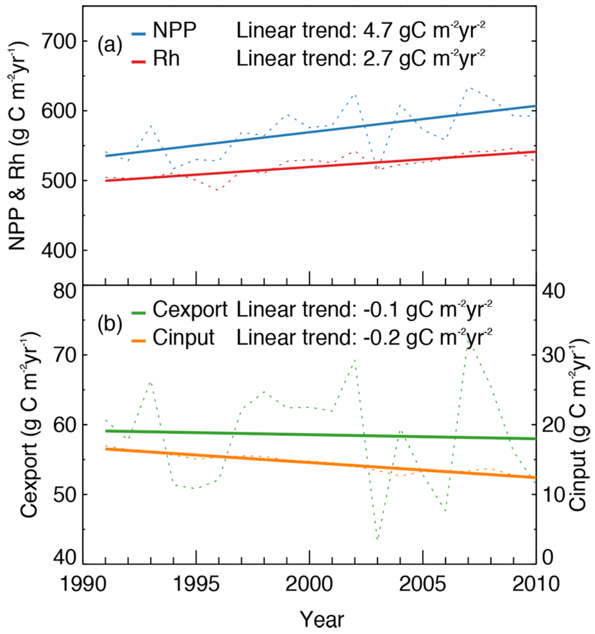
conversion efficiency (section '*Simulation set-up*' and Supporting Information Text S1 for detail).

Figure 4. The spatial distribution of linear trends in NBP during the period 1991-2010 derived from simulation Set-1 considering constant ruminant diet composition and feed conversion efficiency.

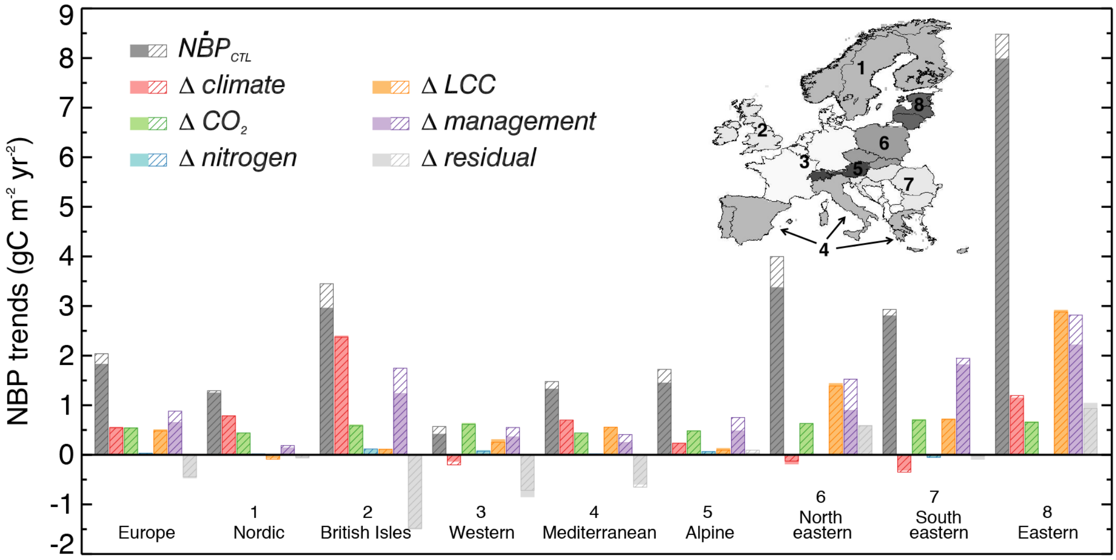
Figure 5. Spatial distribution of the trends in NBP due to: (a) climate change, (b) changes in nitrogen fertilization, (c) changes in grassland area, and (d) in grassland management intensity. Grassland management intensity in this study is given by the fraction of extensively versus intensively managed grasslands; the transition between them (i.e., changes in grassland management intensity) is constrained by the total forage requirement of grass-fed livestock numbers (see Chang *et al*., 2015b for detailed). Results are derived from simulation Set-1 considering constant ruminant diet composition and feed conversion efficiency.

Figure 6. Spatial distribution of the trends in: (a and b) NEP and (c and d) C_{export} due to (a and c) climate change and (b and d) changes in grassland management intensity. Grassland management intensity in this study is given by the fraction of extensively versus intensively managed grasslands; the transition between them (i.e., changes in grassland management intensity) is constrained by the total forage requirement of grass-fed livestock numbers (see Chang *et al*., 2015b for details). Results are derived from simulation Set-1 considering constant ruminant diet composition and feed conversion efficiency.

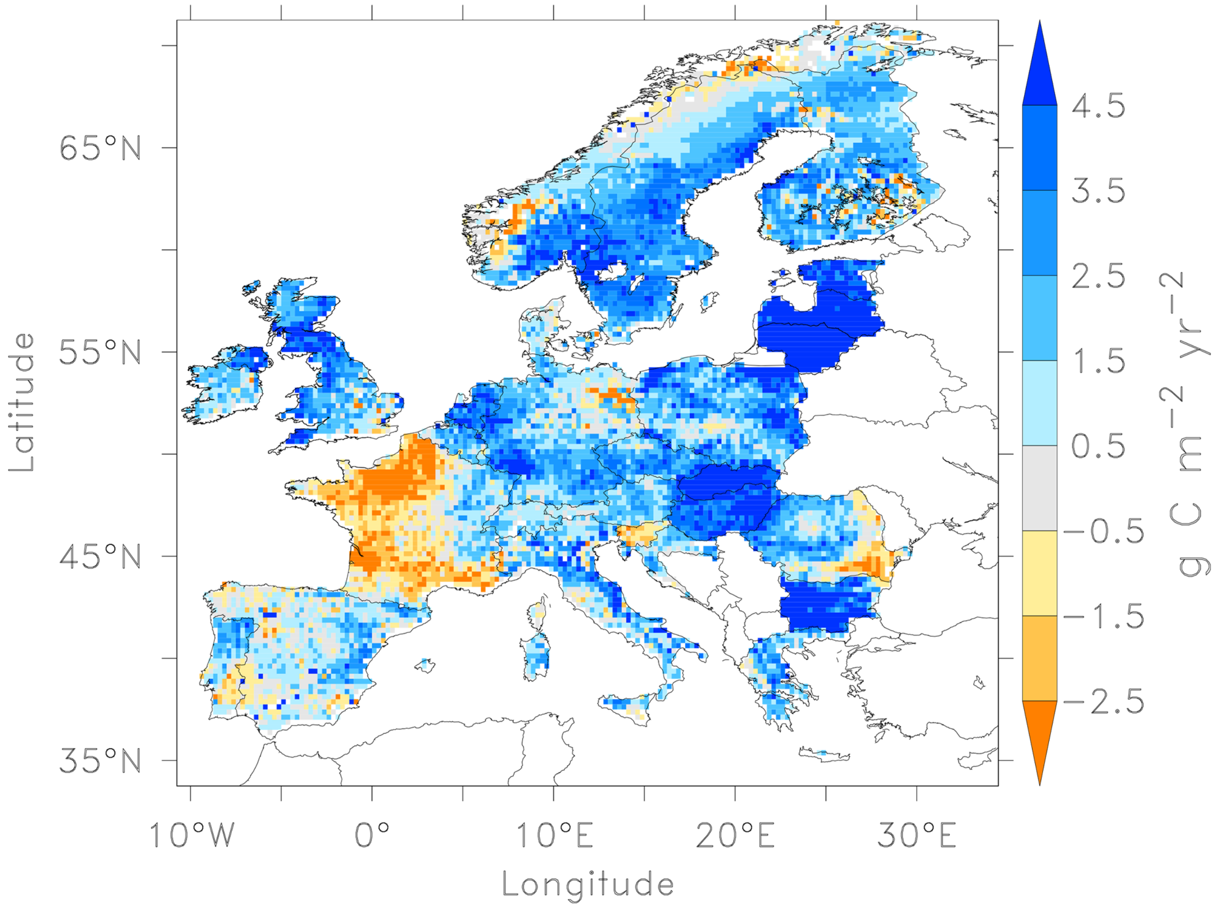
	Climate forcing	Atmospheric CO_2 concentration	Nitrogen addition	Grassland area	Grassland management	NBP trend
E_{CTL}	1991-2010	1991-2010	1991-2010	1991-2010	1991-2010	$N\dot{B}P_{CTL}$
E1	1991-1995	1991-2010	1991-2010	1991-2010	1991-2010	$N\dot{B}P_{climate}$
E2	1991-2010	1991	1991-2010	1991-2010	1991-2010	$N\dot{B}P_{CO2}$
E3	1991-2010	1991-2010	1991	1991-2010	1991-2010	$N\dot{B}P_{nitrogen}$
E4	1991-2010	1991-2010	1991-2010	1991	1991-2010	$N\dot{B}P_{LCC}$
E5	1991-2010	1991-2010	1991-2010	1991-2010	1991	$N\dot{B}P_{management}$

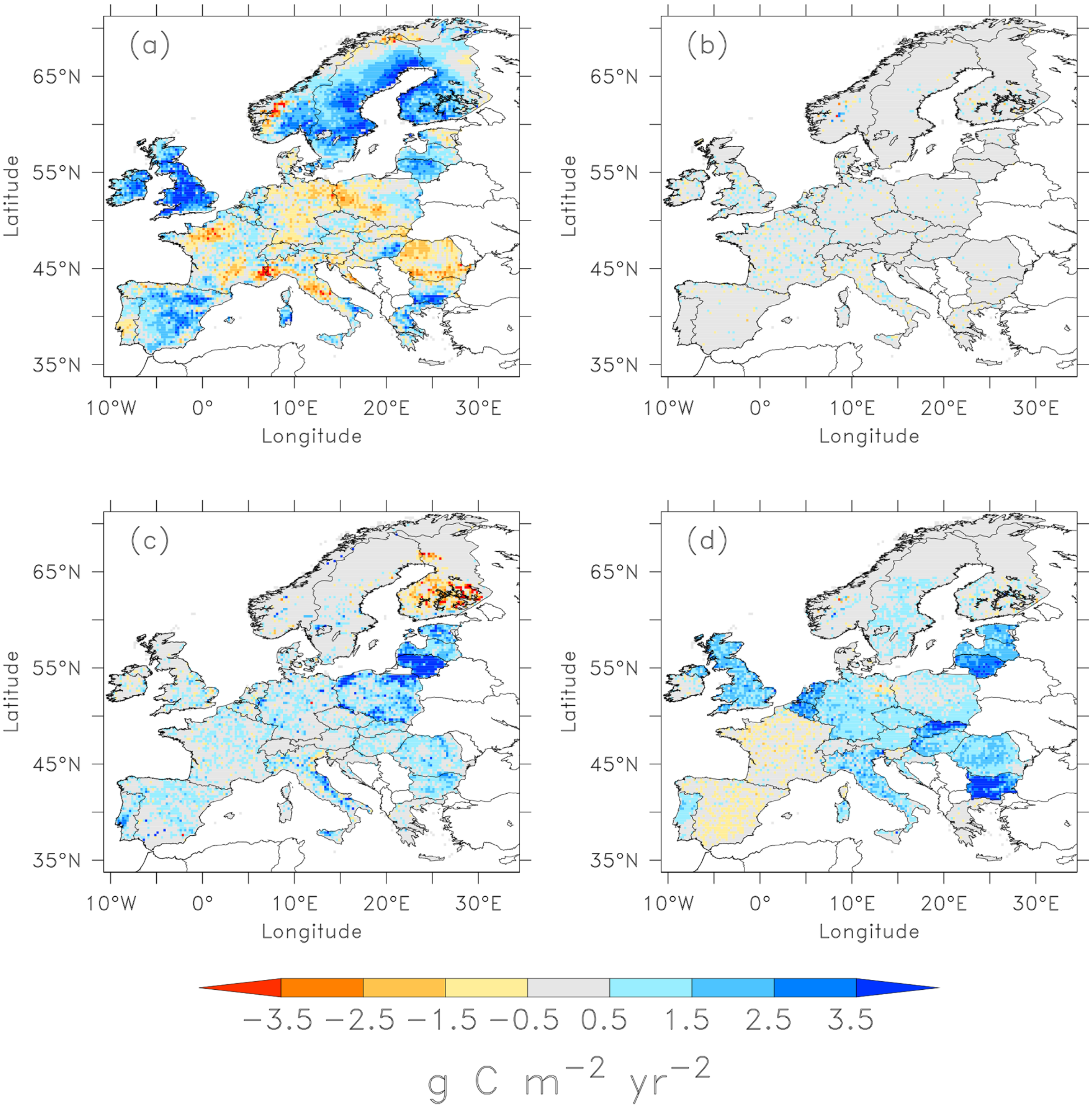

(a)
NPP
Linear trend: 4.7 gC m-2yr-2
Rh
Linear trend: 2.7 gC m-2yr-2
NPP & Rh (g C m-2yr-1)
700
600
500
400
(b)
Cexport
Linear trend: -0.1 gC m-2yr-2
Cinput
Linear trend: -0.2 gC m-2yr-2
Cexport (g C m-2yr-1)
80
70
60
50
40
Cinput (g C m-2yr-1)
40
30
20
10
0
1990
1995
2000
2005
2010
Year

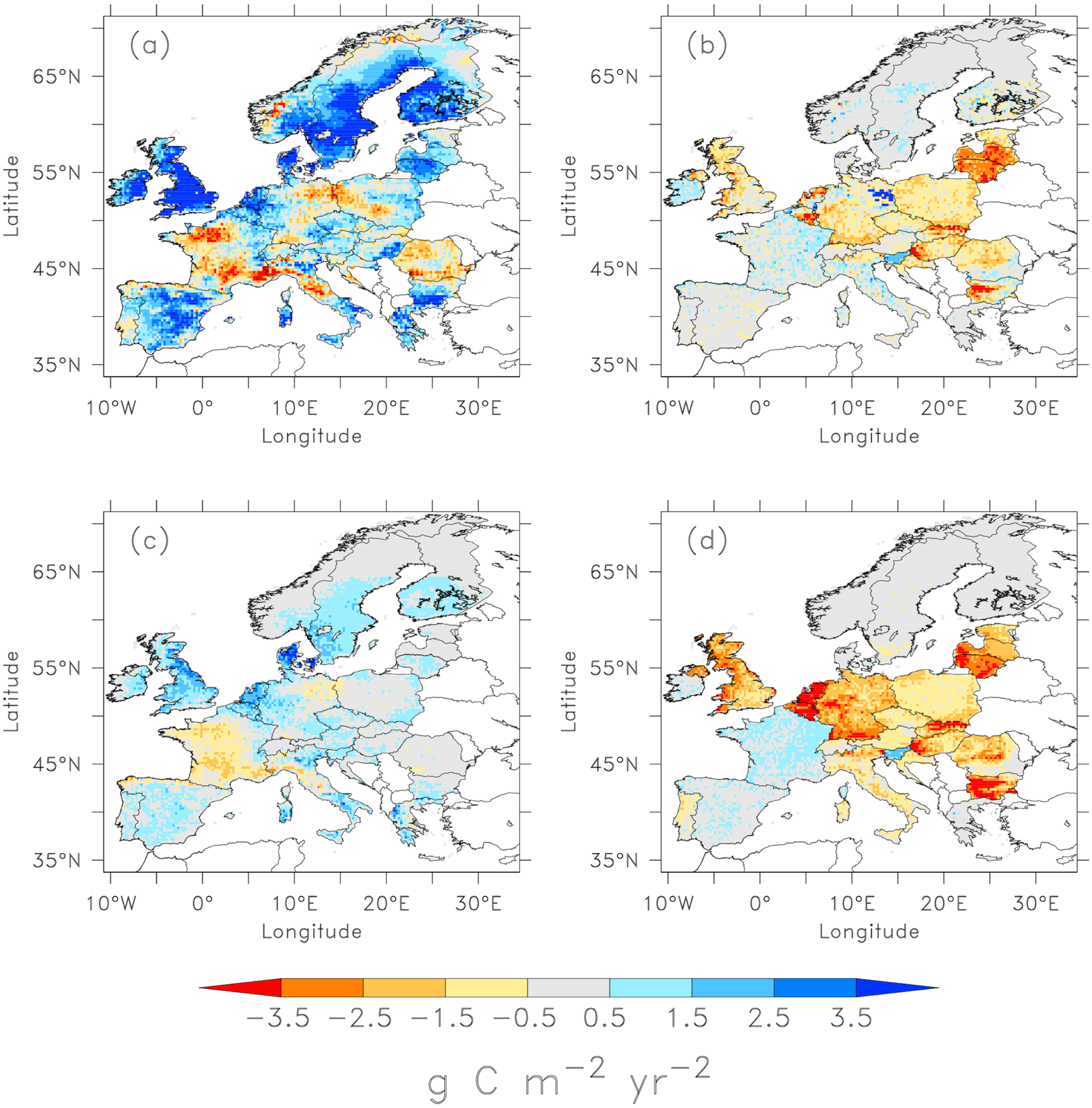
876



NBP trends (gC m-2 yr-2)
NḂP CTL
Δ climate
Δ CO2
Δ nitrogen
Δ LCC
Δ management
Δ residual
Europe
1 Nordic
2 British Isles
3 Western
4 Mediterranean
5 Alpine
6 North eastern
7 South eastern
8 Eastern
9
8
7
6
5
4
3
2
1
0
-1
-2


65°N
55°N
45°N
35°N
Latitude
10°W
0°
10°E
20°E
30°E
Longitude
4.5
3.5
2.5
1.5
0.5
−0.5
−1.5
−2.5
g C m^{-2} yr^{-2}


(a)
(b)
(c)
(d)
65°N
55°N
45°N
35°N
10°W
0°
10°E
20°E
30°E
Longitude
Latitude
−3.5 −2.5 −1.5 −0.5 0.5 1.5 2.5 3.5
g C m^{-2} yr^{-2}


(a)
(b)
(c)
(d)
65°N
55°N
45°N
35°N
Latitude
10°W
0°
10°E
20°E
30°E
Longitude
−3.5 −2.5 −1.5 −0.5 0.5 1.5 2.5 3.5
g C m−2 yr−2