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Effects of land use intensity on the full greenhouse gas balance in an Atlantic peat bog

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Abstract

The assessment of emission factors for many peatlands is difficult, and reliable data on the exchange of carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) between soil and atmosphere of these areas is particularly scarce. Reasons for this are the multitude of soil and land use combinations that control greenhouse gas exchange and the high effort associated with data acquisition.

We investigated the greenhouse gas exchange of a peat bog restoration sequence over a period of 2 yr (July 2007–June 2009) in an Atlantic raised bog in Northwest Germany. We set up three sites representing different land use intensities: intensive grassland (mineral fertilizer, cattle manure and 4–5 cuts per year); extensive grassland (no fertilizer or manure, maximal 1 cutting per year); near-natural peat bog (almost no anthropogenic influence).

We obtained seasonal and annual estimates of greenhouse gas exchange based on closed chamber measurements. CH_4 and N_2O fluxes were recorded bi-weekly, CO_2 NEE determinations were carried out 3–4 weekly. To get annual sums the CH_4 and N_2O fluxes were interpolated linearly while NEE was modelled. The intensive grassland site emitted $548 \pm 169\,g\,CO_2\text{-}C\,m^{-2}$ in the first and $817 \pm 140\,g\,CO_2\text{-}C\,m^{-2}$ in the second year. The extensive grassland site showed a slight uptake in the first year ($-148 \pm 143\,g\,CO_2\text{-}C\,m^{-2}$), and a small emission of $88 \pm 146\,g\,CO_2\text{-}C\,m^{-2}$ in the second year. In contrast to these agriculturally used sites, the near-natural site took up CO_2-C in both years ($-8 \pm 68\,g\,CO_2\text{-}C\,m^{-2}$ and $-127 \pm 53\,g\,CO_2\text{-}C\,m^{-2}$). Under consideration of N_2O and CH_4 exchange, the total average greenhouse warming potential (GWP) for 2008 amounts to $441 \pm 157\,g\,m^{-2}$, $14 \pm 152\,g\,m^{-2}$ and $31 \pm 68\,g\,m^{-2}$ CO_2-C-equivalent for the intensive grassland, the extensive grassland and the near-natural site, respectively.

Despite inter-annual variability, rewetting contributes considerably to mitigating GHG emission from formerly drained peatlands. Already extensively used grassland on moderately drained peat approaches the carbon sequestration potential of near-natural

sites, albeit it may oscillate between being a small sink and being a small source depending on interannual climatic variability.

1 Introduction

The drainage of peatlands for agricultural purposes often causes an increase of CO_2 (carbon dioxide) emissions (Maljanen et al., 2001) and leads to incomplete denitrification due to aerobic conditions with a concomitant rise of nitrous oxide (N_2O) emissions. N_2O emissions are further increased by the use of fertilizers and manure (Jassal et al., 2011; Maljanen et al., 2010a) on drained peatlands. In contrast, the restoration (i.e., rewetting) of drained peatlands can increase the emission of methane (CH_4) (Wilson et al., 2009; Saarnio et al., 2009), nearly up to the level of natural peatlands (Tuittila et al., 2000) or even far above (Hargreaves and Fowler, 1998; Laine et al., 2007). Hence, it is important to consider both effects and to find a trade-off between land use and restoration to reach an optimal greenhouse gas balance in these areas. The mitigation potential for greenhouse gas emissions by changing from intensive to extensive grassland use has been a topic of controversial debates for several years (cf. Robertson et al., 2000; Dalal et al., 2007; Schils et al., 2008 and others). Especially in combination with beneficial effects regarding nature conservation and other ecosystem functions, peat land restoration may have huge potential for reaching internationally agreed sustainability goals (Gorham and Rochefort, 2003; Zedler and Kercher, 2005; Rochefort and Lode, 2006 and others).

For the boreal zone, several studies have demonstrated the contribution of drained and natural peatlands to the atmospheric CO_2 concentration (Kettunen et al., 1999; Dinsmore et al., 2009; Drösler et al., 2008; Ojanen et al., 2010 and others). For the temperate zone information about this contribution is still scarce (e.g., Hendricks et al., 2007; Wilson et al., 2007; Couwenberg, 2011). Furthermore, greenhouse gas (ghg) emissions vary greatly between years (Jungkunst et al., 2006; Iqbal et al., 2009). Thus,

BGD
9, 6793–6828, 2012
Land use effects on GHG balance
S. Beetz et al.

investigations covering more than one year are crucial to provide reliable proxy data for extrapolating ghg emissions to the regional scale.

The total CO_2-C exchange of soils can be simplified into two contrary components: gross primary production (GPP) and soil respiration (R_{ECO}). The main driving force for GPP is the photon flux density of the photosynthetically active radiation or if viewed from the plants perspective the light use efficiency of the plants (cf. Hall and Rao, 1999) while R_{ECO} is controlled mainly by soil temperature (Lloyd and Taylor, 1994) and soil moisture i.e. depth of the water table (Drösler et al., 2008). The lower the water table the deeper the aerated soil zone and consequently the higher the overall intensity of organic matter decay.

In contrast, methane emissions increase with rising water tables because CH_4 is mainly produced by methanogenic bacteria which require anaerobic conditions (Dalal and Allen, 2008). Additionally, CH_4 production depends on temperature (Bellisario et al., 1999; Blodau, 2002). Thus the rewetting of deeply drained grassland sites should aim for a balance between production of CO_2 and CH_4, which seems to be highest at ground water levels around -5 cm (Jungkunst et al., 2008).

N_2O develops in natural peatlands as a by-product of both nitrification and denitrification (Kasimir-Klemedtsson et al., 1997), but emission rates are generally low compared to agricultural land. N_2O is missing from many ghg balance studies because conceptual models of peat land biogeochemistry are based on N-poor bogs (Teh et al., 2011). Managed peatlands often have enhanced N-pools and cycling rates due to fertilization or manuring. Therefore, they have an increased potential for N_2O emissions. Several studies address the N_2O exchange from managed peatlands (see Jungkunst and Fiedler, 2007). Agriculturally used peatlands release from 0.7 up to 3.1 g N_2O m^{-2} yr^{-1} (Flessa et al., 1998) from manure and fertilizers (Velthof and Oenema, 1995).

Full ghg balances have been derived mainly for boreal peatlands (Alm et al., 1999, 2007; Kettunen et al., 1999; Maljanen et al., 2001, 2010b; Ojanen et al., 2011 and others). In the temperate zone, full ghg balances based on the determination of the exchange of all three major ghg between soil and atmosphere are still rare (e.g., Hendriks

BGD
9, 6793–6828, 2012

et al., 2007). The available studies combine eddy covariance and closed chamber measurements but do not account for different land use intensities. Here, we provide full greenhouse gas balances of raised peatlands with different degrees of land use intensity (1: intensively used, 2: extensively used, 3: undisturbed near-natural), based on 2 yr of CO_2, CH_4, and N_2O closed chamber measurements in a peat bog complex in Northern Germany.

We hypothesize that extensive grassland use on moderately drained peatlands leads to a decrease in global warming potential (GWP) compared to intensive grassland use on non-rewetted peat lands. CO_2 emissions are expected to decrease because of the raised water table and hence a less deep aerated soil layer (acrotelm). Because of the same reasons CH_4 emissions are expected to increase. The N_2O emissions should decrease close to a near natural level because no fertilizer and manure were applied.

2 Material and methods

2.1 Site description

The study area is located approximately 80 km northwest of Hamburg at 53° 41′ latitude and 8° 49′ longitude in the peat bog complex "Ahlen-Falkenberger Moor", about 20 km from the North Sea coast (cf. Fig. 1). The climate is humid Atlantic with an average annual precipitation of 925.7 mm and an average annual temperature of 8.5 °C (reference period 1961–1990; DWD 2010). Under these conditions, natural soil formation processes lead to fens and peat bogs in poorly drained areas (Schneekloth, 1981).

The Ahlen-Falkenberger Moor is one of the largest peat bog complexes in Lower Saxony between the estuaries of the Elbe and Weser rivers. In many parts the peat bog complex has been drained for peat extraction since the late 17th century and cultivated for intensive grassland use since the 1950s. Currently, about 60 % of the whole area is used as grassland. A small part of the peat bog complex (approx. 5 %), located in its centre, has never been drained or cultivated and thus remained natural peat bog until today (Höper 2007). In this area vegetation is dominated by cross-leaved heath (*Erica tetralix* L.), flat-topped bog-moss (*Sphagnum fallax* Klinggr.) and common cottongrass (*Eriophorum angustifolium* Honck.). Peat depths range from 330 in cultivated areas up to 515 cm in uncultivated, near-natural areas. The peat in the cultivated areas contains 5 % cotton grass and 1 % heather remnants in the upper layer and is strongly humified down to 15 cm and poorly humified down to 140 cm depth (cf. Table 1). Atmospheric reactive nitrogen (N) deposition is approximately 2.2 to 2.5 $g\,m^{-2}\,yr^{-1}$ (Schröder et al., 2011).

2.2 GHG measurements and gas flux calculation

Within the Ahlen-Falkenberger Moor CO_2, CH_4 and N_2O exchange was measured at three sites with differing land use intensity (Table 1). The intensive grassland (GI) site is managed by 4–5 cuts per year and both mineral fertilization (117.45 $kg\,N\,ha^{-1}$ in 2008, 120.6 $kg\,N\,ha^{-1}$ in 2009) and manure application (226 $g\,C\,m^{-2}$ in 2008 (2.3 $t\,C\,ha^{-1}$), 206 $g\,C\,m^{-2}$ in 2009 (2.1 $t\,C\,ha^{-1}$)). The extensively used grassland site (GE) is not manured and fertilized and only cut once per year. The site has been rewetted in the years 2003/2004. The natural wetland (NW) site is located in a nature reserve area without any drainage or land cultivation. At each site three square PVC collars (0.75 m × 0.75 m × 0.15 m) were installed permanently close to each other. A boardwalk was installed to avoid disturbances when measuring. The positions of the plots were chosen to represent the site characteristics regarding vegetation and site conditions (Fig. 1).

CO_2 exchange was determined from July 2007 through June 2009 at intervals of 3 to 4 weeks. In total we conducted 29 measurement campaigns during these two years. We used square, closed chambers (0.78 m × 0.78 m × 0.5 m) in through-flow (dynamic) mode. Opaque and transparent chambers were placed in turn following the method of Drösler (2005) to obtain data on autotrophic and heterotrophic respiration of the ecosystem (R_{ECO}) and net ecosystem exchange (NEE). Measurements started at sunrise and continued until late afternoon, when soil temperature at 5 cm depth generally reached its maximum value. Following this procedure, the largest possible daily range of the main drivers of CO_2 exchange – photosynthetically active radiation (PAR) and soil temperature – was captured. CO_2 concentrations within the chambers were determined with an infrared gas analyzer (IRGA; LI-820™, Licor®, Lincoln, NE, USA). To prevent heating and to ensure a thorough mixing of air in the chamber headspace, the transparent chamber was equipped with two computer fans that ran continuously during measurements. Additionally, freezer packs were positioned on a frame inside the chamber. With this cooling system, heating of the chamber during measurements was less than 1.5 °C + outside air temperature.

We measured CH_4/N_2O fluxes bi-weekly from July 2007 through June 2009 using opaque chambers. We mixed the air inside the chamber by flushing with a 60 ml syringe shortly before gas sampling, which took place 0, 20, 40 and 60 min following chamber closure. The samples were immediately transferred to an evacuated, airtight, custom-made 20 ml glass vial (Hassa, Lübeck, Germany). The gas analysis was done using a gas chromatograph (Finnigan Trace GC Ultra with Finnigan Valve Oven Trace GC Ultra, Thermo Fisher Corp.) equipped with a Flame Ionization Detector for CH_4 analysis and an Electron Capture Detector for analysis of N_2O concentrations. The precision of analysis for CH_4 and N_2O was 3 to 4 % and 4 % to 5 %, respectively, as determined by replicate injections of calibration gas with ambient concentrations of N_2O and CH_4.

2.3 Environmental parameters

For monitoring of the groundwater table nylon-coated tubes with a diameter of 5 cm perforated in their lower half were installed at each plot. The tubes were equipped with filter slots and a cap at the bottom to prevent water discharge. We measured the water level every 2 weeks with an electric contact gauge. Photosynthetically active radiation (PAR), air temperature, air humidity, precipitation and soil temperature at 5 cm depth were monitored half-hourly by a climate station located close to the three sites.

2.4 Data analysis

Gas fluxes were calculated using the slope of gas concentrations over time:

$$F = k\frac{273.15}{T}\frac{V}{A}\frac{\Delta c}{\Delta t} \tag{1}$$

with F the calculated flux (mg CO_2-C m^{-2} h^{-1}, µg CH_4-C m^{-2} h^{-1} or µg N_2O-N m^{-2} h^{-1}), k a unit conversion factor for calculating fluxes (0.536 µg C µl^{-1} for CH_4 and CO_2 and 1.25 µg N µl^{-1} for N_2O, cf. Flessa et al., 1998), T the mean temperature inside the chamber (K), V the total volume of the chamber in m^3, A the area of the collar (0.5625 m^2), and $\Delta c \Delta t^{-1}$ the concentration change in the chamber headspace over time.

Net ecosystem exchange (NEE) is composed of gross primary production (GPP) and the respiration of the ecosystem (R_{ECO}). We converted the simple equation of Chapin et al. (2006) to calculate NEE:

$$\mathrm{NEE} = \mathrm{GPP} + R_{ECO} \tag{2}$$

There are different methods for modelling R_{ECO} (cf. Maljanen et al., 2001; Wilson et al., 2007). As we did not find any significant relationship between water table and R_{ECO} we used the Arrhenius type model of Lloyd and Taylor (1994) (Eq. 3) to estimate the parameters R_{ref} and E_0 for each measurement campaign.

$$R_{ECO} = R_{ref} \exp\left\{E_0\left[\frac{T_{ref} - T_0}{T_{soil5} - T_0}\right]\right\} \tag{3}$$

with R_{ECO} the measured ecosystem respiration rate (mg CO_2-C m^{-2} h^{-1}), R_{ref} the respiration at reference temperature (mg CO_2-C m^{-2} h^{-1}), E_0 an activation like parameter (K), T_{ref} the reference temperature (283.15 K), T_0 the temperature constant for the start of biological processes (227.13 K), and T_{soil5} the soil temperature in 5 cm depth. Using the campaign specific parameters R_{ref} and E_0 we estimated R_{ECO} at the times of the

BGD
9, 6793–6828, 2012

NEE chamber measurements and subtracted it from the measured NEE values. When it was not possible to calculate a significant relationship between R_{ECO} and T_{soil5} with the data of one measuring day we pooled the data of two measuring days to establish a significant relationship that allowed for the fitting of the R_{ECO} model.

The relationship between the uptake of CO_2 by plants (GPP) and photosynthetically active radiation (PAR) can be modelled using the Michaelis-Menten kinetic (Michaelis and Menten, 1913) and is known to vary greatly between plant species and individual plant development stages (Hall and Rao, 1999). Therefore, we estimated the parameters α (initial slope of the regression curve) and GP_{max} (highest possible production rate at infinite PAR in $mg\,CO_2\text{-}C\,m^{-2}\,h^{-1}$) for each measurement location per measurement date using Eq. (4)

$$GPP = \frac{GP_{max} \cdot \alpha PAR}{GP_{max} + \alpha PAR} \tag{4}$$

with PAR the photon flux density of the photosynthetically active radiation ($\mu mol\,m^{-2}\,s^{-1}$). We then used these parameters to estimate half hourly GPP values. Between measurement dates the plant biomass develops and we assumed a linear development of the model parameters α and GP_{max} between campaigns. Additionally, we set them back to -0.0001 and $-0.01\,mg\,CO_2\text{-}C\,m^{-2}\,h^{-1}$ when the vegetation was cut to represent the loss of green biomass that sets back the ability of the plants to take up CO_2 from the atmosphere. The needed half hourly values of PAR (for modelling GPP) and T_{soil5} (for modelling R_{ECO}) were derived by regression analysis of data from the climate station against data measured on plots per campaign. The resulting relationships were used to obtain PAR and T_{soil5} values per site with half hourly resolution. Using the modelled GPP and R_{ECO} values we calculated half hourly NEE values using Eq. (2).

All our estimations are based on locally fitted models. Therefore, it occurred occasionally that the explaining variables PAR and T_{soil5} took on values that were far out of the model range, which led to unrealistic estimations for R_{ECO} and, thus, GPP in

BGD
9, 6793–6828, 2012

some instances. Therefore, we detected outliers in the daily sums of R_{ECO} and GPP and removed unrealistic values from the dataset. To do so, we grouped the dataset into growing and non-growing season depending on the temperature sums of our climate station (following Janssens, 2010). Then we calculated the interquartile range (IQR) of every subset and removed all values > 1.5 × IQR of the higher quartile and < 1.5 × IQR of the lower quartile (Tukey, 1977). The resulting data gaps were filled by linear interpolation between the marginal data points enclosing the gap. All statistical analyses were done using the software package R (R Development Core Team, 2011).

CH_4/N_2O fluxes per site were calculated together with their standard deviation of the three replicates. Due to a lack of correspondence of water table and soil temperature to CH_4 and N_2O efflux we used linear interpolation between single measurements and integrated the average fluxes per site per year to obtain annual emission sums.

Global warming potentials were calculated using the 2007 IPCC standards (Forster et al., 2007) with a radiative forcing factor of 25 for CH_4 and 298 for N_2O related to CO_2 and a time horizon of 100 yr. Many studies cover only one year (e.g., Hendriks et al., 2007) and often this is not a standard calendar year (e.g., Veenendahl et al., 2007; Lund et al., 2007). However, due to interannual variability, different integration periods may lead to considerable differences in the derived annual budgets. To evaluate the influence of the integration period on the estimated global warming potential we calculated annual budgets using a 365-day shifting window beginning with 1 July 2007 until 1 July 2008.

2.5 Uncertainty analysis

For the locally fitted R_{ECO} and GPP regression models on each measurement day we calculated the standard errors and linearly interpolated the magnitude of the previously calculated standard errors on a half-hourly basis. The total uncertainty of the annual emission sums then was obtained by addition of the temporally weighted errors between measurement days using the law of error propagation.

BGD
9, 6793–6828, 2012

3 Results

3.1 Weather conditions

During the study period mean annual air temperatures were higher (10.2 °C in 2007/2008 and 2008/2009 and 10.8 °C in 2008) and precipitation was slightly lower in the first period (916 mm in 2007/2008) and slightly higher in the second period (929 mm in 2008/2009), and even higher in 2008 (1024 mm) than the long-term averages (8.5 °C / 926 mm). Thus, atmospheric conditions in both years deviated from the long-term (1961–1990) climatic averages (cf. Fig. 2).

3.2 Carbon dioxide exchange

The highest daily C fixation rates (GPP) occurred during July on all sites (Fig. 3). On the intensively used grassland site the highest CO_2-C uptake was modelled for 24 July 2008 ($-16.2 \pm 1.8\,g\,m^{-2}$). On the other two sites the highest CO_2-C uptakes were modelled for July 2007. On the extensively used grassland site as well as on the near-natural site it occurred on 8 July 2007 ($-10.8 \pm 2.5\,g\,m^{-2}$ and $-6.1 \pm 1.2\,g\,m^{-2}$, respectively). According to the models the highest ecosystem respiration rates occurred during July 2007 (16 to 19) on all sites: GI = $20 \pm 0.8\,g\,m^{-2}$; GE = $7.7 \pm 0.9\,g\,m^{-2}$; NW = $6.7 \pm 0.4\,g\,m^{-2}$.

The GI site was the biggest source of CO_2-C in both years with $548 \pm 169\,g\,m^{-2}$ and $817 \pm 140\,g\,m^{-2}$, respectively, while the GE site alternated between being a source and being neutrally with an annual CO_2-C exchange of $-148 \pm 143\,g\,m^{-2}$ in the first and $88 \pm 146\,g\,m^{-2}$ in the second year. In contrast, the NW site acted neutral in this first and accumulated CO_2-C in the second year with $-8 \pm 68\,g\,m^{-2}$ and $-127 \pm 53\,g\,m^{-2}$ (cf. Fig. 3, Table 2). In the calendar year 2008 only the GI site emitted $441 \pm 157\,g\,CO_2$-C on a net basis. Both others sites acted more or less neutrally with 14 ± 152 at the GE site, and $31 \pm 68\,g\,CO_2\text{-C}\,m^{-2}$ at the NW site, respectively.

3.3 Methane exchange

Annual fluxes on the GI site were small with 0.1 ± 0.4 g CH_4-C m^{-2} in 2007/2008 and 0.1 ± 0.1 g CH_4-C m^{-2} in 2008/2009 (Table 2). The GE site emitted little more with 2.6 ± 1.5 g CH_4-C m^{-2} in 2007/2008 and 0.3 ± 0.1 g CH_4-C m^{-2} in 2008/2009. The highest annual emissions of CH_4were detected at the NW site with 5.9 ± 1.2 g CH_4-C m^{-2} in 2007/2008 and 2.8 ± 0.3 g CH_4-C m^{-2} in 2008/2009. Generally we were not able to establish significant relationships between the estimated hourly methane flux rates and typical environmental parameters like soil temperature and water table. On all sites methane emissions were higher in the first year of study compared to the second year (Fig. 4). The hourly methane emissions as well as annual methane emission sums were highest at the NW site followed by the GE site. The GI site exhibited the lowest methane emissions.

The NW site showed a seasonal emission pattern with peaks in the beginning of autumn in 2007 and 2008 as well as in spring 2008. However, emissions did not peak in spring 2009. In contrast, the GI site exhibited clear positive as well as negative peaks only during the winter 2007/2008, whereas the emissions from the GE site showed two relatively distinct peaks (compared to the otherwise low to zero fluxes from that site) in July 2007 and continuous release of methane through winter 2007/2008.

3.4 Nitrous oxide exchange

N_2O fluxes (Fig. 4) showed high temporal variability at all sites. The GI site had the highest N_2O emissions with 112 ± 78 mg N_2O-N m^{-2} in 2007/2008 and 255 ± 101 mg N_2O-N m^{-2} in 2008/2009. The magnitude of N_2O fluxes at the GE site ranged between those of the other two sites, with 42 ± 34 mg N_2O-N m^{-2} in 2007/2008 and 31 ± 24 mg N_2O-N m^{-2} in 2008/2009. The NW site tended to release 9 ± 11 mg N_2O-N m^{-2} in the first and 7 ± 24 mg N_2O-N m^{-2} in the second year.

BGD
9, 6793–6828, 2012

Land use effects on GHG balance

S. Beetz et al.

Generally N_2O fluxes exhibited an erratic emission pattern and could not be related to environmental parameters. We did not detect any seasonality of N_2O fluxes at the NW site. No N_2O emissions were recorded until the summer of 2008. In contrast, the GI site released some N_2O in the beginning of September 2007 as well as in summer 2008 and winter/spring 2009. A similar pattern was found at the GE site with an emission peak in October 2007, and some smaller ones in summer 2008 as well as winter/spring 2009 (Fig. 4).

3.5 Greenhouse warming potential

The GWP of the sites decreased with decreasing anthropogenic impact (i.e., GI > GE > NW, cf. Table 3). Shifting the annual period for integrating the annual GWP (Fig. 5) leads to a considerable variation in annual GWP depending on the temporal determination of the annual integration period (Table 2, Fig. 5). This is most apparent for the GI site where the average GWP of $858 \pm 141\,g\,m^{-2}$ for the period 2008/2009 almost doubles the average GWP of $441 \pm 157\,g\,m^{-2}$ for the calendar year 2008. For the other two sites, shifting the integration period leads to shifts from being a source to being a sink for all three major greenhouse gases (Table 2, Fig. 5).

4 Discussion

4.1 CO_2

The drained, intensively used grassland site (GI site) emitted $434 \pm 157\,g\,CO_2\text{-}C\,m^{-2}$ in 2008. Generally, this is in line with recent findings. For instance, Veenendahl et al. (2007) reported $\sim 420\,g\,CO_2\text{-}C\,m^{-2}\,yr^{-1}$ from intensively used peat grassland in The Netherlands. Couwenberg et al. (2011) summarize several studies and come up with annual emission rates of $410\text{–}760\,g\,CO_2\text{-}C\,m^{-2}$ from temperate grassland. The total emissions as well as the interannual change of the GI site can be explained by differences in land use intensity: the higher frequency of cutting lead to much higher emission rates in the second (5 cuts) than in the first year (3 cuts; Table 3). However, taking into account NEE alone, both years do not differ significantly in their annual CO_2-C exchange (significance of difference of means = 0.33). These findings are in line with Schmitt et al. (2010), who investigated a similarly treated mountain grassland and state that land use and management have a large impact on NEE.

Another indication for the effect of anthropogenic influence is the time until the site becomes a net CO_2 sink again after biomass removal by cutting (Wohlfahrt et al., 2008). When biomass is removed from the site, overall leaf area and consequently GPP decreases substantially (Schmitt et al., 2010). On average, the GI site started to accumulate carbon 22 days after cutting. The GI site was cut 8 times during the whole measuring period, leading to CO_2 net emissions at 176 days or approximately 40 % of the 437 growing season days in total. Therefore, the GPP of the site was mainly controlled by the cutting regime. Additionally, there were 12 manuring events on the site that possibly had an influence on GPP because of covering or contaminating the leaves. Presumably, most of the applied manure is respired quickly (Veenendahl et al., 2007). It is not known how long this effect lasts. Since we did not increase sampling intensity following manure application, we were not able to detect possible increases in CO_2 net release following manuring.

The rewetted, extensively used grassland site (GE site) changed from a small sink in the first to a small source of CO_2 in the second measuring year. This difference was likely caused by the cutting that was carried out on the site once during the study period on 1 October 2008 (Fig. 3). Following cutting GPP was reset to near zero but R_{ECO} was similar in both years which in the sum lead to a significant difference in the total CO_2-C-exchange (NEE) based on this one event.

It is not clear, whether abandoned or rewetted peatlands act as carbon sources or sinks. Hendriks et al. (2007) reported a sink with $-311 \pm 66\,\mathrm{g\,CO_2\text{-}C\,m^{-2}\,yr^{-1}}$ as an average between 2004 and 2006 from an abandoned peat meadow in The Netherlands whereas several studies found abandoned peatlands to be carbon sources (Jacobs et al., 2007, for grassland on organic soils in The Netherlands: $220 \pm 90\,\mathrm{g\,CO_2\text{-}C\,m^{-2}}$, Maljanen et al., 2010b, in Finland: 90 g CO_2-C m^{-2}). The rewetted GE site oscillated between being a source in the first year (−148 ± 143 g CO_2-C m^{-2}) and being neutrally in the second year (88 ± 146 g CO_2-C m^{-2}). Keeping the uncertainties in mind, only long-term measurements can give clarity whether the site accumulates CO_2-C or not. Given that the site is still in progress of restoration, relatively strong fluctuations can be expected because of more pronounced small-scale site differentiation (Drösler et al., 2008). However, compared to the intensively used grassland (GI) site the decrease in NEE is already huge (cf. Fig. 4).

For the natural site (NW site) we calculated a slight accumulation with −14 ± 68 g CO_2-C m^{-2} in 2008 (cf. Table 2). According to Saarnio et al. (2007), natural boreal peat bogs store 48 to 80 g C m^{-2} yr^{-1}. Lund et al. (2007) report C storage rates of 21.5 ± 5.4 g C m^{-2} yr^{-1} from a temperate peat bog in Southern Sweden. Although our near-natural site acted as a CO_2 sink in both years, CO_2-C-uptake increased significantly in the second year compared to the first year, despite a lower GPP because the latter was overcompensated by even stronger decreases in R_{ECO} during the growing season. In winter, R_{ECO} also decreased, but not strongly enough to compensate summertime C gains. This led to a significantly higher annual NEE in the second year.

R_{ECO} depends on the position of the water table because this drives the extent of the aerobic zone in which oxidation occurs, which then influences the C mineralization rate (e.g., Blodau, 2002). Nevertheless, the relationship between R_{ECO} and soil temperature explains most of the variation between summer and winter in both years. All in all, our findings approve the CO_2 sink function of natural peat bogs (cf. Byrne et al., 2004; Drösler et al., 2008, Figs. 3, 6).

4.2 CH_4 fluxes

On the GI site, CH_4-C emissions were generally low – presumably due to the relatively low water table especially during summer. Under low water tables methanogenic processes are suppressed, whereas methanotrophic processes gain importance

(Langeveld et al., 1997). Deeply drained peat bogs can even react as atmospheric CH_4-C sinks (cf. Maljanen, 2003). Thus, the main source for CH_4-C emissions remaining should be manuring which is known to cause great temporal and spatial variation of methane fluxes (cf. Flessa and Beese, 2000). Still, unlike Chadwick et al. (2000) or Augustin (2001), we did not find any peaks after manuring. Most of these peaks occur immediately (6–48 h) after the application (cf. Sherlock et al., 2002; Rohde et al., 2006). Because we did not conduct additional measurements shortly after manuring and strictly followed our biweekly cycle of measurements we might have missed those peaks. The relatively high total CH_4-C emissions at the GE site were likely caused by the raised water level (cf. Dalal and Allen, 2008). Hendriks et al. (2007) reported similar effects from an abandoned peat meadow in The Netherlands but the annual CH_4-C emission rates were about 8 times higher than ours ($19 \pm 16\,g\,CH_4\text{-}C\,m^{-2}$ in 2005 and $15 \pm 12\,g\,CH_4\text{-}C\,m^{-2}$ in 2006 linearly interpolated from a site with water table from 0–40 cm). These relatively high emissions at the Dutch site are likely caused by the more eutrophic soil characteristics.

The highest CH_4-C fluxes as well as the highest variabilities in CH_4-C fluxes were found at the NW site. This is in line with ranges reported in other studies. In a review, Saarnio et al. (2007) reported annual methane emissions from 0.2 up to $16.4\,g\,CH_4\text{-}C\,m^{-2}$ with a mean of $4.6 \pm 4\,g\,m^{-2}$ from 26 pristine peat bogs in Finland. Höper et al. (2008) indicated $19.4\,g\,CH_4\text{-}C\,m^{-2}$ from a natural peat bog in Southern Germany. Methane emissions are often characterized by clear seasonal patterns (e.g., Flessa et al., 1998; Borken and Beese, 2006). Since methanogenic bacteria can only survive in anoxic conditions, the depth of the aerobic zone determines to a large degree the emission of CH_4, and, therefore, water table depth is in general the most important single variable controlling CH_4 fluxes (Roulet et al., 1992; Tuittila et al., 2000). Consequently, we found the highest emissions in the winter seasons of both years when water levels were higher than during the summer season (cf. Fig. 4).

BGD
9, 6793–6828, 2012

Land use effects on GHG balance

S. Beetz et al.

4.3 N_2O fluxes

Although the GI site was the biggest source of N_2O, the annual sum of emissions ($39 \pm 12\,mg\,m^{-2}\,yr^{-1}$) was low compared to similar studies (cf. Byrne et al., 2004: $1\,g\,N_2O$-$N\,m^{-2}\,yr^{-1}$; Alm, 1999: $1.8\,g\,m^{-2}$). At the GE and NW sites, N_2O flux rates were very low and characterized by an erratic pattern which is typical for nutrient poor bog habitats due to the absence of nitrification and low rate of denitrification (Urban et al., 2011). Thus, despite the generally small fluxes, also N_2O emissions decrease from the GI over the GE to the NW site.

For the GI site we assume that only a small part of the fertilizer was transformed into N_2O. Kaiser et al. (1998), for example, found a relative loss from fertilizer to N_2O of 0.7–4.1 % on a loamy silt with winter wheat. Velthof et al. (1995) reported 3.9 % loss for a peat soil under grassland. Related to a total fertilization rate of $11.7\,g\,m^{-2}$ in the first and $12.1\,g\,m^{-2}$ in the second year, the N_2O emissions from GI are in line with these findings (1.0 % in the first and 2.1 % in the second year). Unlike others (e.g., Chadwick et al., 2000; Augustin, 2001; Rodhe et al., 2006) we detected no significant peaks after fertilizer applications. Thus, it is likely that a part of the N_2O emissions derives from the mineralization of organic substance triggered by weather conditions. For example, from January to April 2009 we detected a slight superficial ground frost at the climate station although soil temperature did not fall below zero. Hence, superficial freeze-thaw cycles (cf. Christensen and Christensen, 1991; Flessa et al., 1998; Teepe et al., 2001) may explain the higher emissions during that time.

All N_2O budgets based on temporal upscaling of momentary observations bear the risk of possibly missing of N_2O emission peaks originating by fertilizer or manure application. However, even very short measurement intervals do not guarantee that all emission peaks are recorded because of the extremely high temporal (Kaiser et al., 1998) and spatial (cf. Folorunso and Rolston, 1984; Glatzel et al., 2008) variation of N_2O fluxes. Furthermore, if we really missed peak fluxes, the difference between the intensive site and the other sites is even higher than reported here. This would even

BGD
9, 6793–6828, 2012

strengthen the main point of our argumentation – therefore, the estimate we give is rather conservative.

4.4 GWP

For the year 2008 we did find a gradient of global warming potential with decreasing land use. The GI site still was the biggest source, followed by the natural and the extensive used one as smaller sources. Indeed, GWP not only depended on the intensity of land use and therefore the depth of drainage, but also on the time horizon which was considered for calculating the potential.

We investigated two years of measurements from July to June. In many other studies, different boundaries to separate annual study periods were used. For example, Hendriks et al. (2007) used regular calendar years, Veenendahl et al. (2007) calculated the annual GWP based on data from October to September, and Lund et al. (2007) used data of a period from August to July. Other studies used only the months of the growing season to estimate annual emissions (cf. Tuittila et al., 2000, 2004; Kivimäki et al., 2008; Teh et al., 2011). Using different periods for estimating the global warming potential can yield considerable differences in the reported annual value (Fig. 5). Especially at the GI site emissions of C are apparently lower when considering only the regular calendar year 2008, compared with other averaging intervals. Hence, the range of emissions rises up to more than +100 % of the total emission. Similar results are reported by Lafleur et al. (2003), who found that an ombrotrophic peatland in Canada was a significantly smaller CO_2 sink in a drier year compared to the wetter years before. However, the weather conditions at our site did not change significantly over the two years. After all, our study shows that it is very important to use more than one year for calculating GWP (cf. Drösler et al., 2008), especially when keeping in mind that the values could be used by others to extrapolate to regional estimates.

Furthermore, all three sites showed a significant trend over the time horizon of two years. On the one hand, both sites showed a significant positive trend which is more

BGD
9, 6793–6828, 2012

pronounced at the GI site. The NW site, on the other hand, showed an altering potential from 57 to $-107\,\mathrm{g\,m^{-2}}$ with a significant negative trend (Fig. 5).

4.5 Limitations

A review of the goodness of fit of our models can bee seen in Fig. 6. Despite of the high accuracy there are some limitations regarding our estimate. First, we used only soil temperature for modeling R_{ECO}. There are studies that show significant relationships with other parameters like soil moisture or water table (cf. Wilson et al., 2007), but since we found no improvement of fit when using this parameter, we used the simpler model (cf. Drösler, 2005).

Second, we interpolated the modeling parameters linearly between measurement dates for which they have been derived by fitting the models against the measured fluxes. With this approach fluxes may be overestimated because forage plants initially grow slower directly after cutting. After that initial phase growth rates increase linearly until the genetic capacity is reached (Horrocks and Valentine, 1999). However, in most studies the model parameters are determined using measurements from several field days and therefore the parameters are then used for modeling of GPP for much longer time spans. Here, we used relatively short time steps (3–4 weeks) to represent the changing driving parameters through cutting, manuring or changing weather conditions. The higher the frequency of field measurements the more flexible is the modeling regarding the adaption to changing environmental parameters.

Another aspect is the temperature range and therefore the time span that is used to model R_{ECO}. Other studies refer to year-round measurements (cf. Ojanen et al., 2010) to model R_{ECO}, which increases the fit of the model but decreases the response to the environmental drivers. We tried to create one model for each measuring day. However, the smaller the temperature range for any given day the more difficult is the calculation of significant modeling parameters. To avoid this problem, we pooled data from some of the winter measurements to increase the range of the included temperatures and the reliability of the models.

Discussion Paper | Discussion Paper | Discussion Paper | Discussion Paper |

BGD

9, 6793–6828, 2012

Land use effects on GHG balance

S. Beetz et al.

Finally, we did not consider lateral losses of dissolved gases. By considering these amounts the loss of C can even be higher than found here (Hendriks et al., 2007 found 20.6 ± 4.3 g dissolved $C\,m^{-2}\,yr^{-1}$ outgoing in water from an abandoned peatland in The Netherlands), especially at intensively used grassland sites. Still the carbon leaking this way from the system is usually relative small compared to the gaseous losses (cf. Schulze et al., 2009, who in a review reported $7 \pm 3\,g\,C\,m^{-2}\,yr^{-1}$ loss from grasslands).

5 Conclusions

Only a full greenhouse gas balance allows for the evaluation of the success or failure of restoration measures in terms of climate forcing. Extensivation – in our case the treating of the site with maximum one cut per year and the rise of the water level to approximately 30 cm below ground – can already lead to a considerable reduction of the global warming potential. This is mainly caused by decreased carbon oxidation of the peat due to the higher water level. However, because of the same reason methane emissions increased. Still this reduction can only be seen as a first step. The ultimate goal of restoration measures should be to bring drained and exploited bog peatlands to near-natural conditions, since only under near-natural conditions these areas will be able to accumulate carbon at longer time scales as it was apparent in the GWP trends shown here. Furthermore, the annual GWP values varied considerably depending on the temporal location of the integration period. Therefore, it is crucial to study several years to understand and acknowledge the influence of natural interannual variability. The naturally high interannual variability renders GWP balances based on data series that only cover a year or less highly arbitrary and severely decreases their reliability. Especially in review articles and meta-analyses that bring together many datasets to derive generalisable GWP values for ecosystems or vegetation types comparable integration periods should be used.

BGD
9, 6793–6828, 2012
Land use effects on GHG balance
S. Beetz et al.

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BGD
9, 6793–6828, 2012

Land use effects on GHG balance

S. Beetz et al.

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BGD
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S. Beetz et al.

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BGD

9, 6793–6828, 2012

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BGD
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Land use effects on GHG balance
S. Beetz et al.

Table 1. Soil and land use characteristics of the research sites in the Ahlen-Falkenberger peat bog.

Site	peat depth (cm)	peat state	land use	C/N ratio*	pH	fertilization	Vegetation (dominant species)
GI	330	degraded	intensive grassland	22.2	3.39	mineral fertilizer, cattle manure	*Anthoxanthum odoratum* L., *Lolium perenne* L.
GE	340	degraded	extensive grassland	21.2	3.27	none	*Juncus effusus* L., *Anthoxanthum odoratum* L.
NW	515	near-natural	none	27.7	3.05	none	*Eriophorum angustifolium* Honck, *Sphagnum fallax* Klinggr.

* Displayed is the ratio of the uppermost peat layer, mostly 0–15 cm depth.

BGD
9, 6793–6828, 2012

Land use effects on GHG balance

S. Beetz et al.

Table 2. Components (ecosystem respiration R_{ECO}, gross primary production GPP and net ecosystem exchange NEE) of the global warming potential (GWP) at the sites in different periods. Cutting and manuring are considered.

site/period	R_{ECO} ($g\,m^{-2}$)	GPP ($g\,m^{-2}$)	NEE ($g\,m^{-2}$)	CH_4-C ($mg\,m^{-2}$)	N_2O-N ($mg\,m^{-2}$)	GWP ($g\,m^{-2}$)
GI 2007/2008	2306 ± 68	−1849 ± 153	548 ± 169	146 ± 354	115 ± 78	555 ± 170
GI 2008/2009	2403 ± 78	−1921 ± 114	817 ± 140	73 ± 51	255 ± 101	831 ± 141
GI 2008	2239 ± 71	−1935 ± 137	434 ± 157	248 ± 346	39 ± 12	438 ± 157
GE 2007/2008	1206 ± 43	−1355 ± 136	−148 ± 143	1518 ± 740	43 ± 34	−132 ± 143
GE 2008/2009	1118 ± 48	−1031 ± 137	88 ± 146	261 ± 133	31 ± 24	92 ± 146
GE 2008	1191 ± 45	−1192 ± 145	0 ± 152	1206 ± 555	30 ± 9	12 ± 152
NW 2007/2008	702 ± 28	−709 ± 62	−8 ± 68	5674 ± 978	9 ± 11	44 ± 69
NW 2008/2009	502 ± 19	−629 ± 50	−127 ± 53	2761 ± 255	7 ± 24	−101 ± 54
NW 2008	617 ± 26	−630 ± 62	−14 ± 68	4672 ± 635	17 ± 8	30 ± 68

BGD
9, 6793–6828, 2012
Land use effects on GHG balance
S. Beetz et al.

Table 3. Composition of the total CO_2-C exchange at the intensive grassland site at the Ahlen-Falkenberger peat bog. The number of events is shown in parentheses.

year	NEE ($g\,m^{-2}$)	cuts ($g\,m^{-2}$)	manuring ($g\,m^{-2}$)	total ($g\,m^{-2}$)
$\sum$ 2007/2008	458	317 (3)	226 (7)	549 ± 169
$\sum$ 2008/2009	482	515 (5)	181 (5)	816 ± 140

BGD
9, 6793–6828, 2012

Land use effects on GHG balance

S. Beetz et al.

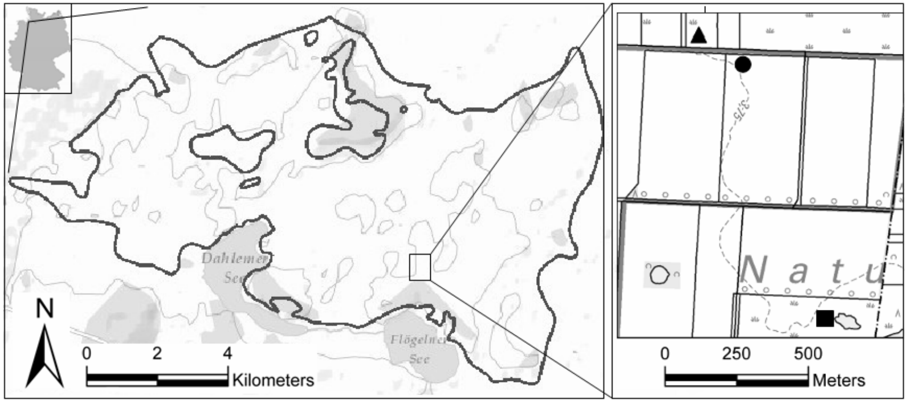

Fig. 1. Location of the Ahlen-Falkenberger peat bog within Germany (upper left small map), and the sites within the inner peat bog complex of the Ahlen-Falkenberger peat bog (left map). On the right map the black triangle (▲) denotes the extensive used site (GE), the circle (•) the intensive used one (GI), and the square (■) the natural one within a natural reserve area (NW), respectively (LGLN, 2012).

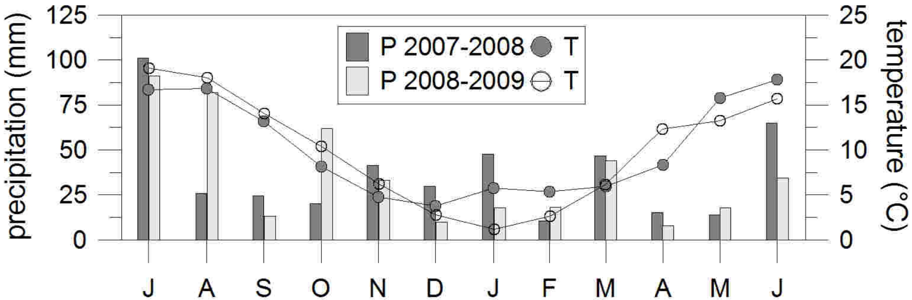

Fig. 2. Monthly values of precipitation and air temperature of both years in comparison. Both years don't differ significantly (permutation test) in climate according to these two variables despite of considerable differences between some months.

Discussion Paper | Discussion Paper | Discussion Paper | Discussion Paper |

BGD
9, 6793–6828, 2012

Land use effects on GHG balance

S. Beetz et al.

Title Page
Abstract | Introduction
Conclusions | References
Tables | Figures
◀| |▶
◀ ▶
Back | Close
Full Screen / Esc
Printer-friendly Version
Interactive Discussion

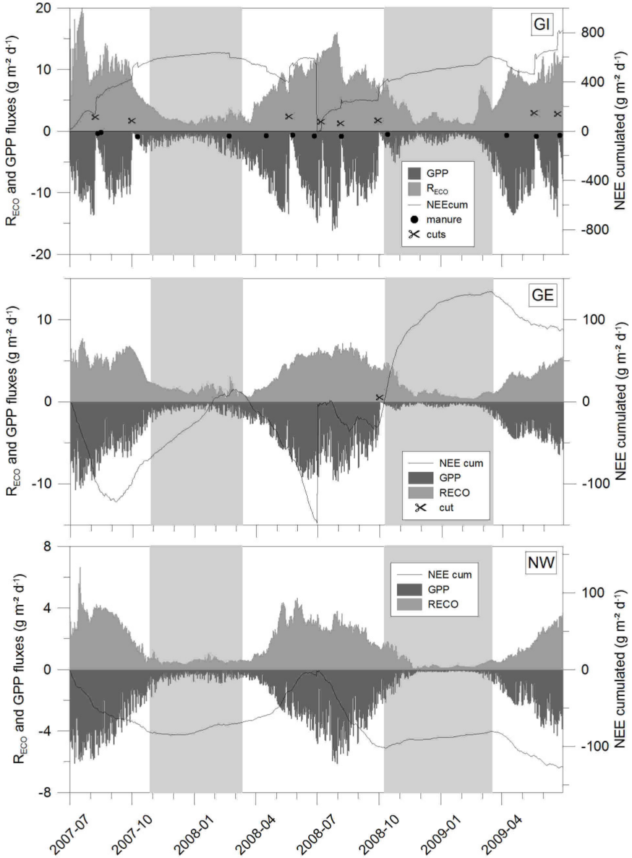

Fig. 3. Modelled CO_2 exchange during 2 yr of measurement. R_{ECO} is above, GPP below zero at the left y-scale. The black line refers to NEE, which is displayed at the right y-scale. White background represents the growing season, grey background non growing season. Note that for the GI site the measures (manuring, cutting) are displayed with respect to their absolute import/export value at the right y-scale.

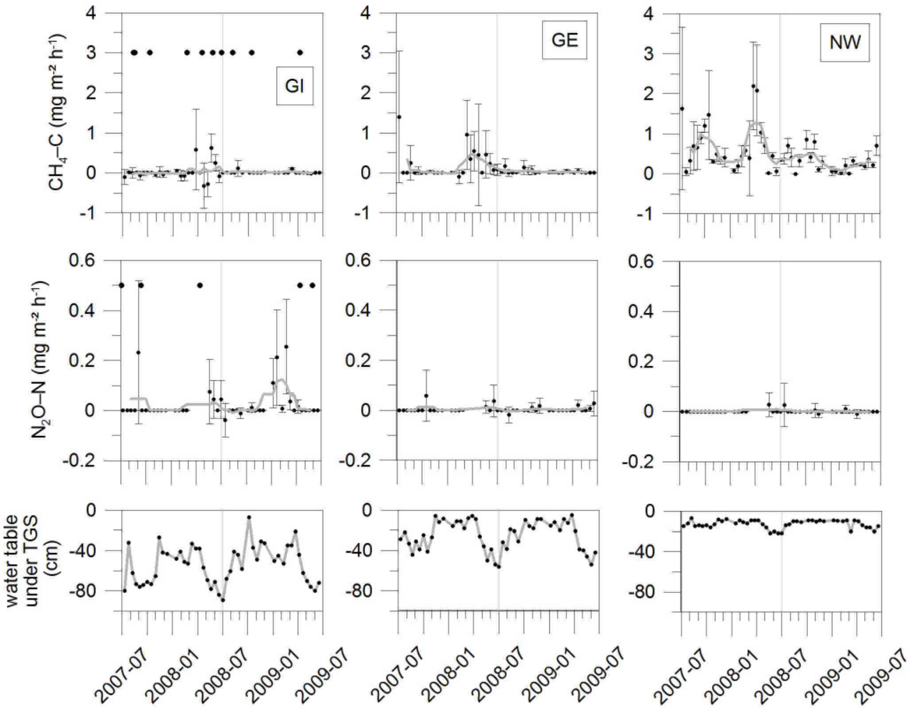

Fig. 4. CH_4-C (top) and N_2O-N-exchange (middle) and water table (down) during the measuring period (from the left to the right the intensive used site GI, the extensive one GE, and the natural peat bog site NW). We had to neglect the N_2O data between 28 February 2008–20 May 2008 because of problems with the gas chromatographer. The grey line symbolizes the running average of 5 values. The black dots in the left column symbolize manuring applications for CH_4 and fertilizer applications for N_2O.

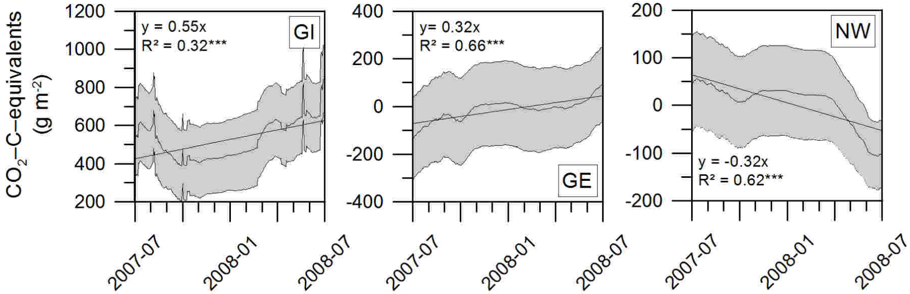

Fig. 5. Annual global warming potential for 365 days with respect to a shifting period of time. x-axis shows the date of beginning of calculation. The shaded area displays the cumulated standard deviation which is calculated as the sum of the daily standard error of NEE and the spatial variation of CH_4/N_2O emission.

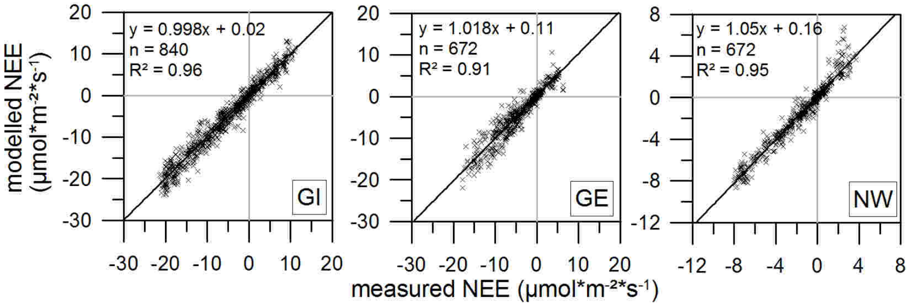

Fig. 6. Scatter plots for NEE model validation. Note the small underestimation at GI and slight overestimation of the GE and NW site.