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The greenhouse gas balance of a drained fen peatland is mainly controlled by land-use rather than soil organic carbon content

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Abstract

Drained organic soils are considered as hotspots for greenhouse gas (GHG) emissions. Particularly arable lands and intensively used grasslands have been regarded as the main producers of carbon dioxide (CO_2) and nitrous oxide (N_2O). However, GHG balances of former peatlands and associated organic soils not considered as peatland according to the definition of the Intergovernmental Panel on Climate Change (IPCC) have not been investigated so far. Therefore, our study addressed the question to what extent the soil organic carbon (SOC) content affects the GHG release of drained organic soils under two different land-use types (arable land and intensively used grassland). Both land-use types were established on a mollic Gleysol (named C_{medium}) as well as on a sapric Histosol (named C_{high}). The two soil types significantly differed in their SOC contents in the topsoil (C_{medium}: 9.4–10.9 % SOC; C_{high}: 16.1–17.2 % SOC). We determined GHG fluxes (CO_2, N_2O and methane (CH_4)) over a period of 2 years. The daily and annual net ecosystem exchange (NEE) of CO_2 was determined with the closed dynamic chamber technique and by modeling the ecosystem respiration (R_{ECO}) and the gross primary production (GPP). N_2O and CH_4 were determined by the close chamber technique. Estimated NEE of CO_2 significantly differed between the two land-use types with lower NEE values (−6 to 1707 g CO_2–C m^{-2} yr^{-1}) at the arable sites and higher values (1354 to 1823 g CO_2–C m^{-2} yr^{-1}) at the grassland sites. No effect on NEE was found regarding the SOC content. Significantly higher annual N_2O exchange rates were observed at the arable sites (0.23–0.86 g N m^{-2} yr^{-1}) compared to the grassland sites (0.12–0.31 g N m^{-2} yr^{-1}). Furthermore, N_2O fluxes from the C_{high} sites significantly exceeded those of the C_{medium} sites. CH_4 fluxes were found to be close to zero at all plots. Estimated global warming potential, calculated for a time horizon of 100 years (GWP_{100}) revealed a very high release of GHGs from all plots ranging from 1837 to 7095 g $CO_{2\ eq.}$ m^{-2} yr^{-1}. Calculated global warming potential (GWP) values did not differ between soil types and partly exceeded the IPCC default emission factors of the Tier 1 approach by far. However, despite being subject to high uncer-

tainties, the results clearly highlight the importance to adjust the IPCC guidelines for organic soils not falling under the definition, to avoid a significant underestimation of GHG emissions in the corresponding sectors of the national climate reporting. Furthermore, the present results revealed that mainly the land-use including the management and not the SOC content is responsible for the height of GHG exchange from intensive farming on drained organic soils.

1 Introduction

Natural peatlands act as a sink for atmospheric carbon dioxide (CO_2) and as a source for methane (CH_4) (Blodau, 2002; Whalen, 2005; Drösler et al., 2008). The net climate effect of natural peatlands regarding the greenhouse gas (GHG) fluxes, however, is close to zero (Drösler et al., 2008). In the last century, drainage and intensification of agriculture turned European peatlands to hot spots for GHG emissions (Drösler et al., 2008). Increased CO_2 and nitrous oxide (N_2O) emissions have been observed from drained peatlands as a result of enhanced decomposition of organic matter (Martikainen et al., 1993; Silvola et al.,1996). The mentioned gases (CO_2, CH_4 and N_2O) act as climatic relevant greenhouse gases (IPCC, 2007). Additionally, N_2O and CH_4 contributes to the chemical destruction of stratospheric ozone (Crutzen, 1979; Solomon, 1999).

Through the ratification of several international agreements on climate protection (e.g. UNFCCC, 1992; Kyoto protocol, 1997 – specified by the Bonn Agreements and Marrakesh Accords, several EU decisions) Germany is obliged to publish annual national greenhouse gas emissions inventories according to the Intergovernmental Panel on Climate Change (IPCC) guidelines. However, the national climate reporting in the Land-use, Land-Use Change and Forestry (LULUCF) sector as well in the Agriculture, Forestry and Other Land-uses (AFOLU) sector is challenging for organic soils. This is mainly because reliable measurements of GHGs from temperate drained peatlands are rare and observed GHG fluxes show a large temporal and spatial variability ranging

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from −2 to 31 $t\,CO_2$–$C\,ha^{-1}\,yr^{-1}$ and 2 to 38 $kg\,N_2O$–$N\,ha^{-1}\,yr^{-1}$ (IPCC, 2014). Furthermore, the definition of histosols is complex (Couwenberg, 2011) and several national and international classification systems exist for organic soils. For the climate reporting under LULUCF/AFOLU, the IPCC guidelines require at least ≥ 10 cm thickness of the soil/peat layer and a C_{org} content of ≥ 12 % in case of a soil thickness ≤ 20 cm for peat soils. Thus, the IPCC definition of peat soils is broader than the definition of histosols in the world reference base for soil resources (WRB, 2008). In the German classification system (KA5) (Ad-hoc-AG Boden, 2005) a distinction is made between soil horizons with ≥ 30 % soil organic matter (SOM) content (called organic horizon) and those, containing 15–30 % SOM (called anmoor horizon). Particularly at the boundary between mineral and organic soils, the conversion from SOM to C_{org} leads to uncertainties due to different conversion factors which are commonly used for mineral soils and peat soils according to the KA5 (Tiemeyer et al., 2013). Depending on the conversion factor (1.72 for mineral soils or 2 for peat soils), the maximum limit of IPCC requirement is between 21 and 24 % SOM (Tiemeyer et al., 2013). Up to date, soils which are, by definition in the transition between mineral and organic soils were mostly neglected in the national GHG inventory of most countries (Leiber-Sauheitl et al., 2014). In the Danish greenhouse gas inventory, for example, GHG emissions from very thin and shallow organic soils, which do not meet the definition of organic soils according to the IPCC, were additionally considered. Due to a lack of information about the release of GHG emissions of those soils, a fixed emission factor, half as much as for typical organic soils (> 12 % C_{org}), has been introduced in Denmark for soils containing 6–12 % organic carbon (Nielsen et al., 2012).

According to estimates, peatlands in Germany account for approximately 5.1 % of the national GHG emissions although they only account for 5.1 % of the total area (NIR, 2010; Drösler et al., 2011). Drained peatlands even represent the largest single source for GHG emissions outside the energy sector in Germany (Drösler et al., 2011; NIR, 2010). Hence, according to the IPCC guidelines, drained peatlands are identified as key category which leads to the fact that Germany is obligated to calculate the annual

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GHG emission inventory on the basis of national specific emission factors (EF; Tier 2 or Tier 3 methods). The main reason for the critical climate balance is caused by the fact that more than two-thirds of the German peatlands are intensively used as grassland or arable land (Drösler et al., 2008). Both land-use types have been regarded as the main producers of CO_2 and N_2O from farmed organic soils (Kasimir-Klemedtsson et al., 1997; Kroeze et al., 1999; Drösler et al., 2008; International Peat Society, 2008). Highest GHG emissions from drained organic soils were related to management activities such as tillage and fertilization which enhance microbial SOM decomposition and nitrogen turnover (Kandel et al., 2013). Beside management practices, several other physical and chemical factors control the intensity of mineralization processes (Heller and Zeitz, 2012) in which soil temperature and soil moisture are considered to be the primary regulators for CO_2 emissions from soils (Silvola et al., 1996; Maljanen et al., 2001; Hardie et al., 2011). However, recent studies have shown that in particular the SOM quality and its labile and more recalcitrant fractions act as key variables affecting the decomposability of SOM and thus control CO_2 fluxes from peatlands (Byrne and Farrell, 2005; Heller and Zeitz, 2012; Leifeld et al., 2012). Beside the macromolecular organic composition (e.g. polysaccharides, lignin, aliphatic biopolymers) of the peat forming vegetation, the SOM quality of peat strongly depends on hydrological and geomorphological building conditions during peat formation (Heller and Zeitz, 2012). Additionally, peat and SOM quality is strongly affected by human impact which leads to peat shrinking, secondary decomposition and mineralization (Heller and Zeitz, 2012). It can be assumed that with increasing peat humification, aggregation and organo-mineral association gain in importance in the SOM stabilization. Thus, a decrease of CO_2 emissions from soils, which are by definition in the transition between mineral soils and peat can be expected compared to peat soils with higher SOM contents. The objective of this study was to quantify GHG emissions from arable lands and grasslands on two types of drained organic soils with different C_{org} contents in South Germany. We hypothesize: (i) that GHG emissions significantly increase with increasing SOC content

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in the soil and (ii) that GHG emissions from arable soils exceed GHG emissions from intensive managed grassland soils.

2 Material and methods

2.1 Study area and experimental design

The study was conducted at a drained fen peatland 30 km north-east of Munich (Freisinger Moos, 48°21′ N, 11°41′ E; 450 m a.s.l.). Since 1914 the Freisinger Moos (FSM) was systematically drained for intensive cultivation (Zehlius-Eckert et al., 2003). Today about 40 % of the whole area is used as grassland and 20 % as arable land (Schober et al., 2008).

According to the climate station in Weihenstephan, located 10 km northeast of the study sites, the 30 yr mean annual temperature was 7.5 °C and the mean annual precipitation was 787 mm (1961–1990). Annual atmospheric N deposition amounted to 6.22 and 7.20 kg N ha^{-1} yr^{-1} in 2010 and 2011. Data of N deposition was collected by the Bavarian State Institute of Forestry at a German Level II monitoring plot (Forest Intensive Monitoring Programme of the UNECE), located in 7 km distance to the investigated sites.

In October 2009, we selected two adjacent areas, one used as intensive grassland and the other as arable land. Both areas are characterized by a distinct gradient in their soil organic carbon (SOC) content in the top soil (Table 1), which increases from southeast to northwest. In March 2010 the arable land was split into two equal halves to simulate two different crop rotations (maize (*Zea mays*) and oat (*Avena sativa*); see Table 2) along the SOC gradient (named A1 and A2). At the grassland area a similar design was conducted to investigate the effect of two different organic fertilizers (named G1, fertilized with cattle slurry and G2, fertilized with biogas digestate). Within these areas we selected two sites with maximum different SOC contents per land-use (Fig. 1). According to the WRB (2006), soil types at the sites were classified as mollic Gleysol

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(named C_{medium}) or as sapric Histosol (named C_{high}) (N. Roßkopf, personal communication, 2010). At each site two plots were selected according to the management type (Fig. 1). A detailed description of the experimental design of the grassland sites and the chemical and physical composition of the applied fertilizers is given in Eickenscheidt et al. (2014b). The arable land was managed according to organic farming criteria but without any fertilization during the investigated period.

At each plot, three PVC-collars for GHG measurements (inside dimension 75 cm × 75 cm) were permanently inserted 10 cm into the soil with a distance of 1.5–2 m to each other. In case of management activities, collars were removed for a short period at the arable land. To prevent oscillations of the peat through movements during the measurements, boardwalks were installed. In March 2010, climate stations were centrally set up between two identical land-use and soil types for the continously recording (every 0.5 h) of air temperature (T_{air}) and humidity at 20 cm above soil surface, soil temperatures at the depth −2, −5 and −10 cm ($ST_{2,5,10}$) and soil moisture content at −5 cm depth. In addition, two further climate stations, additionally equipped with air temperature in 200 cm above soil surface and photosynthetic active radiation (PAR) sensors were operated in close proximity (1.5 km) to the investigated areas. For measuring the groundwater table, plastic perforated tubes (JK-casings DN 50, 60 mm diameter, 1 m length) were inserted close to each collar for plot-specific measurements of groundwater (GW) tables during gas flux measurements at the grassland plots. At the arable land only three tubes were inserted between the two plots of the same soil type. In April 2010, we equipped one tube per plot or, in case of the arable land one tube per soil type, with a water level logger (Type MiniDiver, Schlumberger water services), which recorded the water tables every 15 min. Additionally to the recorded data, plot-specific soil temperatures in three soil depths (−2, −5 and −10 cm) were determined with penetration thermometers at the beginning and end of each gas flux measurement.

2.2 Crop yield, soil sampling and laboratory analyses

Crop yield was determined by harvesting the biomass inside the PVC-collars with a scissor at each harvesting event (same cutting height as the farmers) (Table 2). To determine the annual crop yield, grass samples were oven dried at 60 °C for 48 h and phytomasses of each harvesting event per year were summed. To determine the total carbon (C_{tot}) and total nitrogen (N_{tot}) content, total phytomasses was milled (0.5 mm) and a pooled and homogenized sample from each PVC-collar and harvesting event was analysed by the AGROLAB Labor GmbH (Bruckberg, Germany).

Mineral N ($N_{min} = NH_4^+–N + NO_3^-–N$) contents of each plot were determined according to VDLUFA (1997). Samples were taken during every CH_4/N_2O gas flux measurement. For the determination of C_{tot} and organic carbon (C_{org}), a mixed soil sample of nine individual samples was collected close to each collar at two soil depths (0–10, 10–20 cm) using a 3 cm diameter auger. After drying for 72 h at 40 °C, soil samples were sieved to 2 mm to remove stones and living roots. Analyses were conducted at the Division of Soil Science and Site Science (Humbold Universität zu Berlin, Germany). For the determination of bulk density and porosity, three undisturbed core cutter samples (100 cm^3) were randomly taken at four depths (0–5, 5–10, 10–15, 15–20 cm) for each plot.

2.3 GHG measurements

We measured fluxes of N_2O and CH_4 every second week from December 2009 to January 2012 using the static manual chamber method (Livingston and Hutchinson; 1995). We used opaque chambers (0.78 m × 0.78 m × 0.5 m; PS-plastic, Eching, Germany) which were configured according to Drösler (2005), having two handles at the top, a permanent thermometer for chamber insider temperature (Mini-Thermometer, TFA), a closed cell rubber tube at the bottom to ensure air-tightness when the chamber was positioned on the collars. Furthermore, a vent close to the chamber bottom was connected to a 100 cm PVC tube (4 mm wide) to avoid pressure differences during

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chamber closure and a rubber valve (M20 cable gland, Kleinhuis) for extraction of gas samples was installed at the top of the chamber additionally ensuring pressure release during chamber placement (Elsgaard et al., 2012). In periods when the vegetation grew higher than the chamber height was (0.5 m), extensions were used between the collar and chamber (white, opaque, volume varied between 309 and 1236 L). N_2O and CH_4 gas flux rates were calculated from the linear change in gas concentration over time (four gas samples; 60 min enclosure time, 120 min in case of two or more extensions) considering chamber air temperature and atmospheric pressure. Gas fluxes were accepted when the linear regression was significant ($P \leq 0.05$). In case of small N_2O or CH_4 fluxes, fluxes were also accepted if the coefficient of determination was ≥ 0.90 and the regression slope was between -1 and $1\,\mathrm{ppb\,min^{-1}}$. The cumulative annual mean exchange rate was calculated by linear interpolation between the measurement dates. To minimize diurnal variation in the flux pattern, N_2O and CH_4 sampling was always carried out between 9.00 a.m. and 11.30 a.m. We removed the gas fluxes measured in 2010 from the data set due to errors in the gas chromatography (GC) analysis and due to long vial storage. To improve GC accuracy a methanizer was installed in late 2010. Further, it was ensured that vial storage time did not exceed two weeks in 2011. A detailed description of gas sampling and gas chromatograph settings is given in Eickenscheidt et al. (2014a, b).

For CO_2 flux measurements we used the closed dynamic manual chamber system which was described in detail by Drösler (2005) and Elsgaard et al. (2012). Camber configuration was identical with N_2O/CH_4 chambers as above mentioned. CO_2 measurement campaigns took place in irregular time intervals (8–60 days) depending on weather conditions, management activities and the phenological stage of plants (Table S1–S8 in the Supplement). Measurement campaigns always started one hour before sunrise and lasted till late afternoon to cover the full range of the photosynthetic active radiation (PAR) and air and soil temperature. Opaque and transparent chambers (same dimension as for N_2O and CH_4 measurements) were alternately used at each of the three collars per plot during the time course of a measurement campaign to obtain the ecosystem respiration (R_{ECO}) and the net ecosystem exchange (NEE). In total up to 55 NEE measurements and 33 R_{ECO} measurements were conducted per measurement day and plot (Table S1–S8). As for N_2O and CH_4 measurements, extensions were installed between the collar and chamber in case of vegetation growing higher than the chamber height was (transparent or opaque, volume varied between 309 and 1236 L). Chambers were connected to an infrared gas analyser (IRGA, LI-820, LI-COR, USA), which continuously determined the CO_2 headspace concentration. In the case that extensions were used, chamber air from each level of an extension (every 0.5 m) was sucked and merged to guarantee a reliable mixture signal from inside the chamber. Additionally, three fans (SUNON® Super Silence MAGLev®-Lüfter) continuously operated during the measurement to ensure a constant mixing of the chamber air. Chamber enclosure time was 120 s for transparent chambers and 240 s for opaque chambers, respectively. The CO_2 concentration, air temperature from inside the chamber and site specific PAR was recorded every 5 s with a data logger (GP1 Data logger, Delta-T Devices, UK). To prevent heating of the air in the transparent chambers, freezer packs (1–10 pieces) were positioned in the air stream of the fens at the inner surface of the PVC collar (Drösler, 2005; Beetz et al., 2013). Single measurements where the PAR changed more than 15 % of the starting value or the temperature inside the chamber increased more than 1.5 °C compared to the outside air temperature were discarded and measurement was repeated (Leiber-Sauheitl et al., 2014). CO_2 gas fluxes were calculated by linear regression. Non significant gas fluxes ($P \geq 0.05$) with slopes close to zero or zero (equilibrium between GPP and R_{ECO}) were not discarded (Alm et al., 2007; Leiber-Sauheitl et al., 2014). For NEE flux calculation, a minimum time interval of 25 s was used, whereas for R_{ECO} fluxes a minimum interval of 60 s was applied.

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2.4 Modeling of CO_2 net ecosystem exchange

The net ecosystem exchange (NEE) of CO_2 is defined as the product of the gross primary production (GPP) and the ecosystem respiration (R_{ECO}) (Chapin et al., 2006).

$$\mathrm{NEE} = \mathrm{GPP} + R_{ECO} \tag{1}$$

In the present study we followed the atmospheric sign convention in which a positive NEE is defined as a net flux of CO_2 to the atmosphere (Elsgaard et al., 2012).

2.4.1 Modeling of ecosystem respiration

The measured R_{ECO} fluxes are the sum of autotrophic (R_a) and heterotrophic (R_h) respiration. Both compartments are mainly controlled by temperature (Lloyd and Taylor, 1994; Tjoelker et al., 2001). For each measurement campaign and plot the dependency between R_{ECO} and temperature was modeled according to Lloyd and Taylor (1994) who developed an Arrhenius type relationship to predict soil respiration rates (Eq. 2).

$$R_{ECO} = R_{ref} \cdot e^{E_0 \cdot \left(\frac{1}{T_{ref}-T_0} - \frac{1}{T-T_0}\right)} \tag{2}$$

R_{ECO}	ecosystem respiration [mg CO_2–C m^{-2} h^{-1}]
R_{ref}	respiration at the reference temperature [mg CO_2–C m^{-2} h^{-1}]
E	activation energy [K]
T_{ref}	reference temperature: 283.15 [K]
T	temperature constant for the start of biological processes: 227.13 [K]
T	air or soil temperature [K]

In response to the phenological stage of the plants, management activities or changing soil moisture conditions, the applied temperature as explanatory variable could change during the year. Therefore, the R_{ECO} model was fitted to the appropriate temperature type (air temperature in 20 cm or soil temperature in −2, −5 or −10 cm) which showed

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the best explanatory power for R_{ECO}. In case that the temperature span was too small for model building (e.g. winter time, snow cover) or a significant relationship between the temperature and R_{ECO} could not be observed (e.g. after ploughing), an average CO_2 flux was calculated for the measurement campaign. Annul sums of R_{ECO} were calculated by summing 0.5 hourly R_{ECO} fluxes recalculated from Eq. (2), based on the linear interpolated parameters R_{ref} and E_0 of two consecutive measurement campaigns and the continuous time series of air and soil temperatures (Elsgaard et al., 2012). In case of management events (e.g. harvesting, plugging, etc.) or snow cover, R_{ref} and E_0 were kept constant from the previous measurement campaign until the management date. After the management, parameters were taken from the subsequent measurement campaign (Leiber-Sauheitl et al., 2013). However, in case of harvesting at the grassland plots, estimated parameters were linearly interpolated over this period. Estimated parameters and used temperatures for R_{ECO} are shown in Table S1 to S8.

2.4.2 Modeling of gross primary production

We estimated GPP as the product of measured NEE minus modeled R_{ECO} at the same time step, since it is not possible to determine GPP through measurements. The relationship between GPP and PAR was modeled by a Michaelis–Menten type rectangular hyperbolic function proposed by Falge et al. (2001) (Eq. 3).

$$\mathrm{GPP} = \frac{\alpha \cdot \mathrm{PAR}}{\left(1 - \left(\frac{\mathrm{PAR}}{2000}\right) + \left(\frac{\alpha \cdot \mathrm{PAR}}{\mathrm{GPP}_{2000}}\right)\right)} \tag{3}$$

GPP	gross primary production [mg CO_2–C m^{-2} h^{-1}]
α	initial slope of the curve;
	light use efficiency [mg CO_2–C m^{-2} h^{-1}/µmol m^{-2} s^{-1}]
PAR	photon flux density of the photosynthetic active radiation [µmol m^{-2} s^{-1}]
GPP_{2000}	gross primary production at PAR 2000 [mg CO_2–C m^{-2} h^{-1}]

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Prior to modeling GPP, we corrected the plot specific PAR values since the acrylic glass of the transparent chambers absorbed at least 5 % of the incoming radiation (PS-plasitc, Eching, Germany) (Leiber-Sauheitl et al., 2014). Annual sums of GPP were calculated based on the linear interpolation of α and GPP_{2000} between two consecutive measurement campaigns and the continuous time series of the PAR (Drösler, 2005; Elsgaard et al., 2012). In case of management events (e.g. harvesting, plugging, etc.) α and GPP_{2000} were kept constant from the preceding measurement until the management time and were set to zero at the 0.5 h time step during the working process. Thereafter, parameters were immediately linearly interpolated from the subsequent measurement campaign for the grassland plots. For the arable land plots, parameter interpolation started after the establishment of the seed. Estimated parameters are shown in Tables S1 to S8.

2.4.3 Model evaluation and uncertainties analysis

For R_{ECO} and NEE model evaluation, we used Pearson's correlation coefficient (r), Nash–Sutcliffe efficiency (NSE) (Nash and Sutcliffe, 197), percent bias (PBIAS) and the ratio of the root mean square error to the SD of measured data (RSR) (Moriasi et al., 2007). According to Moriasi et al. (2007) model simulation can be judged as satisfactory if NSE > 0.50 and RSR ≤ 0.70. For PBIAS, the optimal value is 0.0, with low-magnitude values indicating accurate model simulation. Additionally, positive PBIAS values indicate model underestimation bias, and negative values indicate model overestimation bias (Gupta et al., 1999; Moriasi et al., 2007). To account for the uncertainties in annual R_{ECO} and annual GPP modeling, annual sums from the upper and lower limits of the determined parameters (R_{ref}, E_0, α, GPP_{2000}), based on their standard errors (SE) were estimated (Drösler, 2005; Elsgaard et al., 2012). However, quantifying total model uncertainties is challenging because of the multiple sources of errors (Beetz et al., 2013) and due to a lack of independent data for gap-filling verification. The main uncertainty in the present study may derive from management activities where no additional measurements were conducted and parameters were kept constant (e.g. R_{ref} and E_0 at the grassland) or set to zero (e.g. α and GPP_{2000} at the grassland).

2.5 Estimation of NECB and GWP

A simple net ecosystem carbon balance (NECB) was calculated for each plot based on the NEE, the carbon export of harvested phytomass, the carbon input through organic fertilizer application and the cumulative annual CH_4 exchange (Elsgaard et al., 2012; Beetz et al., 2013).

To assess the global warming potential (GWP) from the different plots the net emissions of carbon equivalents of NECB and N_2O were summed according to Beetz et al. (2013). For the conversion of CH_4 and N_2O to CO_2 equivalents, radiative forcing factors of 25 and 298 were used (Forster et al., 2007).

2.6 Statistical analyses

Statistical analyses were conducted using R 3.1 (R Development Core Team, 2013). The assumption of normality of residuals was tested using the Lilliefors or Shapiro–Wilk test and by plotting the Quantile–Quantile plots. Homogeneity of variances of residuals was checked using the Levene or Breusch–Pagan test and by plotting the residuals against the fitted values. Where necessary, data were box-cox transformed prior to analyses. For the comparison of cumulative modeled GPP, R_{ECO} and NEE as well as for annual yields and N_{min} values we used a two-factorial ANOVA with land-use and soil type as fixed effects (including an interaction term in the model), neglecting the individual plot specific standard error for modeled CO_2 values. Non-significant terms were removed from the model structure. In case of significant differences among the means, we used Tukey's honest significant differences (TukeyHSD). For GW level we used the nonparametric Kruskal–Wallis Rank Sum test and the non-parametric Pairwise Wilcoxon Rank Sum test with Bonferroni correction for multiple comparisons. For testing two independent sample means regarding the two investigated years 2010 and

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2011, we use the Welch two sample *t* test (C_{org} contents, bulk density, yields) or the non parametric Mann–Whitney *U* test (for N_{min}). Due to temporal pseudoreplication of time series data (N_2O, CH_4 field measurements) we applied linear mixed effects models (Crawley, 2007; Hahn-Schöfl et al., 2011; Eickenscheidt et al., 2014a and 2014b). For N_2O fluxes we set up a basic model with land-use type and soil type as fixed effects and the spatial replication (individual plot) nested in time as random effect. We extended the basic model by a variance function due to observed heteroscedasticity. Furthermore, N_2O fluxes showed significant serial correlation. To take this into account, a first-order temporal autoregressive function was included in the model. Autocorrelation was tested using the Durbin–Watson test and by plotting the empirical autocorrelation structure. The model extension was proved by the Akaike Information Criterion (AIC). For multiple comparisons we conducted Tukey contrasts using the General Linear Hypotheses function from the "multcomp" package (Hothorn et al., 2013). CH_4 fluxes did not satisfy the necessary requirements for the linear mixed effects model therefore CH_4 analysis were restricted to the nonparametric Mann–Whitney *U* test. We accepted significant differences if $P \leq 0.05$. Results in the text are given as means ±1 SD.

3 Results

3.1 Environmental variables

Temperatures between the two investigated land-use types and soil types did not differ considerably. In 2010 and 2011, air temperature in 20 cm height ranged from −17.5 to 39.5 °C with an annual mean of 8.6 °C at both grassland sites in 2011. Soil temperature in −2 cm soil depth averaged 10.3 °C at the GC_{medium} site and 10.5 °C at the GC_{high} site in 2011. At the arable land air temperature in 20 cm height ranged from −15.0 to 39.5 °C in 2010 and 2011, with an annual mean of 8.8 °C AC_{medium} and 8.7 °C AC_{high} in 2011. Soil temperature in −2 cm soil depth averaged 10.1 °C at both arable land sites in 2011. Longer periods of snow cover occurred in the period 1 January to 12 March

2010, 28 November 2010 to 10 January 2011 and from 24 January to 5 February 2011 (see also Figs. 4 and 5). In 2011, the annual sum of PAR was 17 % higher compared to the year 2010. Annual precipitation amounted to 850 mm (2010) and 841 mm (2011) in the investigated period, which was slightly above the 30 years mean of the period 1961–1990. Mean annual groundwater levels of the C_{high} sites were significantly higher (all $P < 0.001$) compared to the C_{medium} sites in 2010 and 2011 (Table 1). Furthermore the GW level at the arable sites were significantly higher (all $P < 0.001$) compared to the grassland sites in both investigated years. Longer periods of flooding and water saturation were only observed at the AC_{high} sites for the period from 1 to 17 June 2010.

3.2 Soil properties and mineral nitrogen contents

Total organic carbon contents and bulk density in the 0–10 and 10–20 cm soil layers significantly (all $P < 0.01$) differed between the two soil types investigated (Table 1). At the grassland sites pH values in the 0–20 cm soil layer were approximately one unit lower compared to the arable land (Table 1). Observed C/N ratios at the soil depth 0–20 cm were between 1 and 12 (Table 1), indicating nitrogen-rich conditions at all plots. Extractable N_{min} contents of the soils ranged from 1 to 178 mg N kg^{-1} at the arable sites and from 2 to 115 mg N kg^{-1} at the grassland sites (Figs. 2 and 3). In both years, the N_{min} contents at the grassland sites significantly ($P < 0.001$) exceeded those from the arable site (Fig. 3). Furthermore the N_{min} contents of the C_{high} sites were significantly ($P < 0.01$) higher compared to the C_{medium} sites (Fig. 3), but this was not valid considering the arable land separately. Slightly higher N_{min} contents were found at the soil depth 10–20 cm compared to the soil depth 0–10 cm, but differences were only significant for the grassland sites ($P < 0.05$). In both years, N_{min} was mainly dominated by NO_3^-, whereas NH_4^+ was only of minor importance. However, at the AC_{high} sites the proportion of NO_3^- in the soil depth 0–10 cm was lower (approximately 80 %)

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compared to the AC_{medium} sites (approximately 97 %), whereas at the grassland sites no differences were found between the two soil types investigated (91–95 %).

3.3 Biomass yield

The mean annual crop yield ranged from 1.2 ± 0.5 to $10.2 \pm 1.6\,\mathrm{t\,DM\,ha^{-1}\,yr^{-1}}$ at the arable land and from 6.2 ± 0.7 to $13.1 \pm 2.9\,\mathrm{t\,DM\,ha^{-1}\,yr^{-1}}$ at the grassland in 2010 and 2011 (see also Eickenscheidt et al., 2014b) (Table 3). For both land-use types the crop yield was significatly ($P < 0.01$) lower in the year 2010 compared to the year 2011 (73 % at the A sites and 52 % at the G sites). However, it has to be taken into consideration that at the grassland sites three instead of two cuts were carried out in 2011. At the arable land a longer period with partially flooding and high water saturation damaged or partly killed the maize seedlings as well as the oat plants in June 2010, especially at the C_{high} sites. Furthermore, in 2010 the entire plants were harvested and used as silo maize or oat corn and straw respectively, whereas in 2011 only the grains were harvested regarding both management practices and the remaining plants were left on the field. In both years investigated, the yield from the grassland sites significantly exceeded those from the arable land (all $P < 0.001$), whereas no significant differences were found between the two soil types observed.

3.4 CO_2 fluxes

The modeling showed that the air temperature in 20 cm above soil surface and soil temperature in −2 cm are the main drivers of R_{ECO} in the present study, while soil temperatures in −5 and −10 cm soil depth mostly showed distinctly weaker correlations (Tables S1–S8). At the arable land, 88 % of the calculated models based on T_{air}, and only 12 % on ST_2, whereas at the grassland sites 54 % of the models based on T_{air} and 39 % on ST_2. Model evaluation statistics from observed R_{ECO} vs. modeled R_{ECO} generally revealed a good model performance with a slight tendency of model overestimation bias for the year 2010 (mean PBIAS −2.39). Pearson's correlations coefficients for ob-

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served R_{ECO} vs. modeled R_{ECO} ranged between 0.89 and 0.98, NSE values ranged from 0.70 to 0.97 and RSR values were ≤ 0.55 (Table 4). According to the annual temperature trend, R_{ECO} showed a clear seasonality with maximum flux rates during the summer time. In 2010, highest daily R_{ECO} fluxes of up to 41 $g\,CO_2\text{–}C\,m^{-2}\,d^{-1}$ were modeled at the A2C$_{medium}$ (oat) and G1C$_{medium}$ plot, whereas in 2011, distinctly lower maximum daily R_{ECO} fluxes of up to 28 $g\,CO_2\text{–}C\,m^{-2}\,d^{-1}$ and 32 $g\,CO_2\text{–}C\,m^{-2}\,d^{-1}$ were modeled for the A2C$_{high}$ (maize) plot and the G2C$_{high}$ plot, respectively (Figs. 4 and 5). At the grassland sites, annual sums of modeled R_{ECO} ranged from 3521 ± 1041 to 4316 ± 562 $g\,CO_2\text{–}C\,m^{-2}\,yr^{-1}$, which was significantly ($P < 0.001$) higher compared to the arable sites where R_{ECO} ranged from 2012 ± 284 to 2992 ± 230 $g\,CO_2\text{–}C\,m^{-2}\,yr^{-1}$ (Table 3, Fig. 6a). Differences in R_{ECO} between the two soil types investigated were only small and not significantly different (Fig. 6a).

Like R_{ECO}, GPP showed a clear seasonal trend with increasing CO_2 uptake capacity with increasing PAR intensity in summer time. In 2010, highest maximum daily GPP of up to −25 $g\,CO_2\text{–}C\,m^{-2}\,d^{-1}$ were modeled for the arable land (maize, C$_{medium}$) and up to −20 $g\,CO_2\text{–}C\,m^{-2}\,d^{-1}$ for the grassland (G2C$_{high}$), whereas in 2011, distinctly higher GPP values up to −35 $g\,CO_2\text{–}C\,m^{-2}\,d^{-1}$ were modeled for both maize plots and up to −28 $g\,CO_2\text{–}C\,m^{-2}\,d^{-1}$ for the G2C$_{high}$ plot (Figs. 4 and 5). At the grassland sites annual sums of GPP ranged between −2093 ± 152 and −2962 ± 178 $g\,CO_2\text{–}C\,m^{-2}\,yr^{-1}$, which was significantly ($P < 0.01$) higher compared to the arable sites where GPP ranged between −873 ± 110 and −2360 ± 237 $g\,CO_2\text{–}C\,m^{-2}\,yr^{-1}$ (Table 3, Fig. 6b). Differences in GPP between the two soil types were not significant.

Calculated NEE were in good agreement with observed NEE. Nevertheless, the calculated percent bias revealed a tendency of model overestimation for both years (mean PBIAS −7.5 in 2010 and −6.1 in 2011). Pearson's correlations coefficients for observed NEE vs. calculated NEE ranged from 0.79 to 0.98, NSE values ranged from 0.61 to 0.96 (Table 5). The mean RSR values was 0.36. Annual NEE significantly (P < 0.01) differed between the two land-use types with lower NEE values at the arable sites, ranging from

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-6 ± 546 to $1707 \pm 619\,g\,CO_2\text{–}C\,m^{-2}\,yr^{-1}$, compared to the grassland sites were NEE ranged from 1354 ± 740 to $1823 \pm 851\,g\,CO_2\text{–}C\,m^{-2}\,yr^{-1}$ (Table 3, Fig. 6c). Differences between the two soil types were not significant for NEE.

3.5 N_2O and CH_4 fluxes

Nitrous oxide emissions were generally low at all plots (Fig. 7). N_2O fluxes rarely exceeded $50\,\mu g\,N\,m^{-2}\,h^{-1}$. However, single N_2O peaks with maximum flux rates of up to $2832\,\mu g\,N\,m^{-2}\,h^{-1}$ were detected at the 3 June at both maize plots as well as at the 6 September at both oat plots with maximum flux rates of up to $289\,\mu g\,N\,m^{-2}\,h^{-1}$. At the grassland sites, highest N_2O fluxes of up to $992\,\mu g\,N\,m^{-2}\,h^{-1}$ were found immediately after fertilizer application (see Eickenscheidt et al., 2014b). In general, N_2O fluxes from the arable sites were significantly ($P < 0.01$) higher compared to the grassland sites (Fig. 8a). Furthermore, N_2O fluxes from the C_{high} sites significantly ($P < 0.05$) exceeded N_2O fluxes from the C_{medium} sites, but this was not valid considering the arable land separately. Significant differences within the land-use types, regarding N_2O flux rates, were only found between the grassland plots, where the application of biogas digestate significantly ($P < 0.01$) enhanced the N_2O fluxes compared to the application of cattle slurry (see Eickenscheidt et al., 2014b). At the arable land distinctly different N_2O flux rates between maize and oat were not found, but the single peak emissions observed led to significantly ($P < 0.01$) higher annual cumulative N_2O emissions at the maize plots (Table 3, Fig. 8a). N_2O peaks accounted for 75 and 87 % of the annual N_2O balances at the maize plots, whereas at the oat plots peaks account for 63 and 54 % of the annual N_2O sums (at C_{medium} and C_{high}, respectively). Annual cumulative N_2O emissions ranged from $0.12 \pm 0.01\,g\,N\,m^{-2}\,yr^{-1}$ ($G1C_{medium}$) to $0.86 \pm 0.21\,g\,N\,m^{-2}\,yr^{-1}$ ($A2C_{high}$) (Table 3).

Most of the time, all sites showed a weak uptake of CH_4 or zero fluxes. CH_4 peaks up to $173\,\mu g\,C\,m^{-2}\,h^{-1}$ were occasionally found immediately after fertilization at the G1 sites (see Eickenscheidt et al., 2014b). Moreover, a high CH_4 peak event of up to

$2177\,\mu g\,C\,m^{-2}\,h^{-1}$ occurred on the 14 July 2011 at the oat plots. Generally, CH_4 fluxes of the arable sites significantly ($P < 0.01$) exceeded CH_4 fluxes of the grassland sites, whereas no differences were found between the two soil types investigated (Figs. 7 and 8b). Significantly different CH_4 fluxes within the land-use types could not be observed regarding the annual fluxes in 2011. However, considering the annual cumulative exchange rates, CH_4 emissions of the oat plots significantly ($P < 0.05$) exceeded those of the maize plots. The observed weak CH_4 emissions or uptakes amounted to cumulative annual CH_4 exchange rates ranging between $-0.11 \pm 0.05\,g\,C\,m^{-2}\,yr^{-1}$ ($G2C_{medium}$) and $0.51 \pm 0.17\,g\,C\,m^{-2}\,yr^{-1}$ ($A1C_{medium}$) (Table 3). However, as previously mentioned for N_2O, the single CH_4 peak event observed at the arable sites entirely controls the cumulative sum of CH_4 and turns the plots from a sink into a source of CH_4.

3.6 NECB and GWP

Including the C export from harvested phytomass, C import from fertilization and CH_4 exchange to NEE, calculated NECB ranged from 451 ± 617 to $1894 \pm 872\,g\,C\,m^{-2}\,yr^{-1}$. Estimated GWP's ranged from 1837 ± 2293 to $7095 \pm 3243\,g\,CO_{2\,eq.}\,m^{-2}\,yr^{-1}$, revealing a very high release of greenhouse gases from all plots (Table 6). However, CO_2 dominated the GWP of all plot to nearly 100 % (range between 97–99 % and for maize 86–90 %), whereas the contribution of N_2O and CH_4 were almost negligible, with exception of the maize plots.

4 Discussion

4.1 Magnitude of GHG fluxes

The observed annual CO_2 emissions were in the upper range or partly higher than CO_2 exchange rates reported in the literature from temperate drained arable lands

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(e.g. Maljanen et al., 2001, 2007, 2010; Grønlund et al., 2008; Höper et al., 2008; Leifeld et al., 2011; Elsgaard et al., 2012; Drösler et al., 2013) and grasslands (e.g. Maljanen et al., 2001, 2010; Grønlund et al., 2006, 2008; Elsgaard et al., 2012; Beetz et al., 2013; Drösler et al., 2013; Leifeld et al., 2014; Renou-Wilson et al., 2014). No differences in the CO_2 release of the C_{medium} and C_{high} sites were found in the current study, and no information about CO_2 fluxes of comparable soils to those of the C_{medium} sites were available in the literature. Observed CO_2 emissions from the arable land were in the range or partly doubled (4.51–12.04 t CO_2–C ha yr^{-1}) the IPCC default emission factor from the Tier 1 approach for drained boreal and temperate arable lands (7.9 t CO_2–C ha yr^{-1}; IPCC, 2014) whereas more than three times higher CO_2 emissions were observed at the grassland sites (15.81–18.94 t CO_2–C ha yr^{-1}) compared to the IPCC default emission factor for deep-drained temperate grasslands (6.1 t CO_2–C ha yr^{-1}; IPCC, 2014). However, comparison of CO_2 exchange rates is difficult since annual variability is very high. For example Leifeld et al. (2014) reported that the NECB of a temperate grassland in Germany ranged from 0.98 to 19.46 t C ha^{-1} yr^{-1}, with a five year mean of 9.06 ± 6.64 t C ha^{-1} yr^{-1}. In this study the highest value was observed for the period 2010 to 2011 which was in good agreement with the values estimated by us during this period. The finding is also in line with Kasimir-Klemendtsson et al. (1997), who reported net CO_2 exchange rates ranging from 8 to 115 t CO_2 ha^{-1} yr^{-1} for farmed organic soils, demonstrating the high bandwidth of measured CO_2-balances.

Observed cumulative annual N_2O emissions were distinctly lower than the default emission factor from the Tier 1 approach for boreal and temperate, drained arable land (13 kg N_2O–N ha^{-1} yr^{-1}; IPCC, 2014) and for temperate deep drained, nutrient rich grassland (8.2 kg N_2O–N ha^{-1} yr^{-1}; IPCC, 2014). In line with this, several other authors reported much higher N_2O emissions from organic soils ranging from to 61 kg N_2O–N ha^{-1} yr^{-1} for arable lands (Kasimir-Klemendtsson et al., 1997; Augustin et al., 1998; Flessa et al., 1998; Petersen et al., 2012; Drösler et al., 2013) and ranging from 1.15 to 41 kg N_2O–N ha^{-1} yr^{-1} for grasslands (Velthof et al., 1996; Augustin

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et al., 1998; Flessa et al., 1997, 1998; van Beek et al., 2010, 2011; Kroon et al., 2010; Petersen et al., 2012; Beetz et al., 2013; Drösler et al., 2013).

As expected, observed CH_4 fluxes from all plots were low, which is in line with generally low groundwater levels and the absence of aerenchymous plant species which can transport CH_4 from an anaerobic layer to the atmosphere, bypassing the oxic zone at the soil surface (Grosse et al., 1992; Svensson and Sundh, 1992; Whalen, 2005). Cumulative annual CH_4 emissions or uptakes were in the range reported for other deep drained arable lands and grasslands (Maljanen et al., 2010; Petersen et al., 2012; Beetz et al., 2013; Drösler et al., 2013; Renou-Wilson et al., 2014) and fit also well with the IPCC default emission factor for boreal and temperate drained arable land ($0\,kg\,CH_4\,ha^{-1}\,yr^{-1}$; IPCC, 2014). A distinctly higher emission factor however is given by the IPCC for a temperate deep-drained, nutrient-rich grassland ($16\,kg\,CH_4\,ha^{-1}\,yr^{-1}$; IPCC, 2014) compared to our estimations.

4.2 Uncertainties in GHG fluxes and modeling

Several factors probably influenced the accuracy of estimated CO_2 exchange rates. Firstly, the R_{ECO} model based only on temperature changes disregarding the effect of soil moisture or GW level. Thus changing soil moisture contents or GW levels between two consecutive measurements campaigns were neglected since we assume a linear change in derived model parameters (see also Beetz et al., 2013; Leiber-Sauheitl et al., 2014). Secondly, management activities like ploughing at the arable sites probably produced peak CO_2 emissions, which we may have missed. Additionally, it can be assumed that after harvesting at the grassland sites, R_{ECO} decreased due to the reduced phytomass. However, additional measurement campaigns to capture this effect did not take place in the current study and no corresponding data were found in the literature. Furthermore, it is well known that the application of organic fertilizers produced short-term CO_2 emission peaks, which were also not sufficiently detected. However, both sources of errors may even have an opposite effect. Thirdly for GPP, linearly interpolation of parameters produced some uncertainties since it can be as-

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sumed that plant growth after cutting did not increase linearly (Horrocks and Valentine, 1999; Beetz et al., 2013). However with the available data set, it was not possible to quantify the error by the used interpolation approach of parameters since the dataset was too small for cross validation and no additional measurements for an independent model validation were conducted. In addition, despite high model accuracy, the calculated PBIAS revealed a slightly model overestimation bias for R_{ECO} and NEE for both years (R_{ECO} only in 2010). Thus, modeled R_{ECO} and calculated NEE rates should be considered as a conservative estimation. However, modeled values fit well with values reported in the literature (see Fig. 9).

Several studies indicated that dissolved organic C can significantly contribute to terrestrial C balances (e.g. Worrall et al., 2009; Dinsmore et al., 2010 Renou-Wilson et al., 2014). Thus, for the calculation of NECB from drained organic soils, fluvial C losses should additionally be considered in future investigations.

Observed N_2O fluxes showed a high temporal variability with long periods of low background emissions and a few high peaks, mainly after management activities. Measurement frequency was increased after fertilization at the grassland plots for at least two weeks (see Eickenscheidt et al., 2014b) but due to our regular measurement intervals in the remaining year we cannot rule out that we may have missed high N_2O events driven by changing climate conditions (e.g. drying–rain or freeze–thaw events) and/or management activities, particularly at the arable sites. N_2O peaks are known to last a couple of days up to several weeks (Stolk et al., 2011). Due to our measurement intervals and interpolation approach, observed N_2O and CH_4 peaks distinctly altered the cumulative annual budgets, increasing the overall uncertainties in estimated GHG emissions. However, for future investigations in GHG emissions we strongly advocate the combined use of automatic and manual chamber systems to maintain a higher accuracy of data.

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4.3 Soil organic carbon effects

With exception of N_2O, significantly different GHG emissions between the two soil types investigated were not found in the present study, although significantly different SOC contents in the upper soil horizon were detected. The observation is in strong contrast to our hypothesis that GHG emissions significantly increase with increasing SOM content (hypothesis i).

Regarding CO_2 fluxes, the current findings are however in line with investigations from Leiber-Sauheitl et al. (2014), who reported that CO_2 emissions were not related to different SOM contents in the upper horizon of an extensive grassland in North Germany. Contrary, Veenendaal et al. (2007) and Renou-Wilson et al. (2014) assumed that their different estimated respiration rates for grassland sites were driven by different SOC/SOM contents. However, it can be assumed that not only the SOM content itself acts as a key factor controlling the CO_2 release, but the proportion of SOM which is exposed to mineralization, which in turn is driven by drainage depth. Therefore we calculated the effective C stock as the fraction of aerated carbon in the soil profile according to Leiber-Sauheitl et al. (2014) (Fig. 10). No relationship was found between the effective C stock and the C flux components (expressed as NECB), which was also reported by Leiber-Sauheitl et al. (2014) and Tiemeyer et al. (2014). However, Fig. 10 shows that at the grassland sites, C stocks available for mineralization processes are comparable (40–45 $\mathrm{kg\,C\,m^{-2}}$), probably explaining the equal CO_2 loss rates from this land-use type. Temperature and soil moisture are considered to be the primary regulators for CO_2 emissions from soils (Silvola et al., 1996; Maljanen et al., 2001; Hardie et al., 2011), since they directly affect microbial activity and the rate of enzymatic processes (Michaelis and Menten, 1913; Tietema et al., 1992). In the present study, temperatures are found to be equal at all sites due to their close proximity, whereas the soil moisture contents significantly differed between the C_{high} and C_{medium} sites mainly according to the GW oscillation. It is well known that the water level height has a strong influence on CO_2 emissions from peatlands as it directly affects the oxygen

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availability for microbial activity as was reported in several studies (e.g. Silvola et al., 1996; Berglund and Berglund, 2011; Renou-Wilson et al., 2014; Leiber-Sauheitl et al., 2014). Beside abiotic factors substrate chemistry, in particular the SOM quality and its labile and more recalcitrant fractions, are considered to act as key variables affecting the decomposability of SOM and thus controlling CO_2 fluxes from peatlands (Byrne and Farrell, 2005; Heller and Zeitz, 2012; Leifeld et al., 2012). For example, Leifeld et al. (2012) showed that the soil respiration rate of a disturbed temperate peatland was strongly controlled by its polysaccharides content, particularly the O-alkyl-C content was found to be a useful proxy for respiration rates. SOM quality was not examined in our study, but both soil types at all plots investigated exhibited highly decomposed organic material (H10, according to Von Post's humification scale; N. Roßkopf, personal communication, 2013). This is typical for organic soils which have been drained and intensively managed for a long time, and is in line with Leifeld et al. (2012), who found that organic matter quality declines with ongoing decomposition, resulting in low polysaccharides contents and a lower availability for heterotrophic metabolism. Nevertheless, observed NECB revealed very high C loss rates from the SOC pool. Leifeld et al. (2014) suggested that intensive management, drainage and changed climate drivers accelerate peat decomposition today, and therefore outweighed declining peat quality. Additionally, Reiche et al. (2010) reported that the degree of humification is not suitable for the prediction of CO_2 and CH_4 fluxes from anaerobic decomposition, which stands in contrast to assumptions made by Glatzel et al. (2004). However, observed equal narrow C/N ratios (10–12) in the upper soil reveal firstly a high organic matter quality, easily to mineralize, and secondly comparable SOM qualities at all plots, probably explaining why no significantly different C loss rates between the two different soil types were found in the present study.

In line with CO_2, CH_4 fluxes were also not different between the two soil types investigated, but this can mainly be attributed to the intensive drainage and thus soil aeration, which effectively inhibited microbial methanogenesis at the C_{medium} and C_{high} sites. It is known that the availability and quality of organic substrates influences the amount

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of produced CH_4. Nevertheless, several studies indicate that high CH_4 fluxes in bogs are mainly controlled by labile organic substrates such as root exudates or plant litter and not by bulk peat (Minchin and McNaughton, 1984; Chanton et al., 1995; Bridgham et al., 1998; Whalen, 2005; Hahn-Schöfl et al., 2011).

In contrast to CO_2 and CH_4 fluxes, N_2O fluxes from the C_{high} sites significantly exceeded N_2O fluxes from the C_{medium} sites. This can probably be attributed to the more favorable soil conditions for denitrification, supported by higher N_{min} contents and higher groundwater levels at these sites (Eickenscheidt et al., 2014b). In both years N_{min} was mainly dominated by NO_3^-, demonstrating that net nitrification entirely controls net nitrogen mineralization at all plots. Thus, nitrification provided the substrate for denitrification and additionally, may itself have contributed to N_2O production. In general, N_2O production processes are various and can occur simultaneously within close proximity (Davidson et al., 1986; Butterbach-Bahl et al., 2013). Both nitrification as well as denitrification depend on the availability of labile organic compounds as C and/or energy source (Butterbach-Bahl et al., 2013), in which autotrophic nitrification depends particularly on the availability of CO_2 for cell growth (Delwiche and Finstein, 1965). However, for denitrification the actual regulation by C is currently not yet understood (Baggs and Philippot, 2011), but it can be assumed that sufficient metabolizable C was widely available at all plots investigated.

4.4 Land-use and management effects

At peatlands GW level and land-use type are closely linked. From a meta-analysis of 53 German peatlands Tiemeyer et al. (2013) found that the mean annual GW level was lower for arable land than for intensive grassland with median GW levels of approximately −70 and −37 cm above soil surface. The GW levels observed in our study were on average lower at the arable land and higher at the grassland compared with the average of the meta-analysis. In general, intensive farming at peatlands presupposes low GW levels, since most of the arable crops are not adapted to low oxygen contents in the rhizosphere as could be seen in the present study, where the temporarily high

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GW level caused plant damage and yield losses at the arable sites in 2010. The effect of reduced biomass productivity due to high GW levels which inhibited photosynthesis by slowing the rate of gas diffusion through the vegetation (Lohila, 2008) was also reported by Renou-Wilson et al. (2014). Both annual sums of GPP as well as yields were in good agreement with those reported from other peatlands as can be seen in Fig. 9. Statistical analysis revealed significantly higher yields at the grassland sites compared to the arable sites, but it has to be taken into account that at the arable sites only the grains were harvested in 2011 and up to 3.84 and 9.05 $t\,DM\,ha^{-1}$ remained on the field regarding the oat and maize plots, respectively. Due to the continuous plant cover over the whole year at the grassland plots annual sums of GPP were significantly higher at these plots compared to the arable plots in 2010 as well as in 2011.

As GPP, modeled annual sums of R_{ECO} significantly differed between the two land-use types with distinctly higher R_{ECO} values at the grassland sites. As mentioned above, R_{ECO} is strongly controlled by temperature since it stimulates both R_a and R_h, as can be seen in the pronounced seasonality of R_{ECO}. From the model fits it can be suggested that the more frequent model adaptation with T_{air} (88 %) reveals a higher share of R_a at the arable site compared to the grassland sites. At the later, approximately 40 % of the R_{ECO} models based on ST_2, perhaps demonstrating a more balanced ratio of R_a and R_h. Nevertheless, the proportion of the different respiration compartments of R_{ECO} is unknown, but Silvola et al. (1996) reported that root-derived respiration from grasslands established on peatland accounted for 35–45 % of total soil respiration. Furthermore, Maljanen et al. (2001) found that root-associated respiration on grasslands was distinctly higher compared to arable lands. However, the significantly higher R_{ECO} at the grassland sites can firstly perhaps be related to the higher biomass production at these sites, because a higher GPP also results in higher above- and below-ground autotrophic respiration (Leiber-Sauheitl et al., 2014; Renou-Wilson et al., 2014). Moreover, the increased transport of photosynthates to the plant rhizosphere due to the higher GPP may favor bacterial metabolism through increased root exudates (Mounier et al., 2004; Henry et al., 2008; Sey et al., 2010), additionally enhancing R_h. Secondly, the organic fertilizer application at the grassland plots stimulates microbial growth and thus SOM mineralization (Gutser et al., 2005; Jones et al., 2007). Additionally, a large part of the C from the organic fertilizer will quickly be metabolized to CO_2 (Vuichard et al., 2007). Several authors (see e.g. Dao, 1998; Maljanen et al., 2010) reported that regularly ploughed and fertilized arable lands are larger sources of CO_2 than non-tilled arable land soils or grasslands, due to aerating and mixing of crop residues into the soil. However, in the current study the effect of management is difficult to capture.

Despite of higher modeled GPP values, the distinctly higher modeled R_{ECO} values led to significantly higher calculated NEE values at the grassland sites compared to the arable sites. With the exception of the maize plot at the C_{medium} site in the year 2011, all plots show positive NEE balances in both years investigated, as expected for drained organic soils and as commonly reported in the literature (e.g. Maljanen et al., 2001, 2010; Grønlund et al., 2006, 2008; Elsgaard et al., 2012; Beetz et al., 2013; Drösler et al., 2013). However, the huge CO_2 uptake capacity during the short growth period of the maize plants, compensates for the soil CO_2 release due to microbial decomposition of organic matter at least in the year 2011. Nevertheless, as seen in the NECB, the C export also reversed the maize cultivation on the C_{medium} site to a C source. Previous studies of annual NEE from maize on organic soils are rare in literature, but our results are in line with Drösler et al. (2013) who reported NEE values ranging from −216.2 to 443.8 $g\,C\,m^{-2}\,yr^{-1}$. As mentioned above, it has to be take into account that in the year 2011 only the grains were harvested at all arable plots. Assuming that silage maize would have been produced instead of maize grains or the straw was additionally harvested at the oat plots, NECB would partly be doubled and more comparable to calculated grassland values.

According to Maljanen et al. (2010) the better aeration of regularly ploughed arable land leads to a larger sink of atmospheric CH_4 compared to permanent grasslands. This contrasted our results, where the CH_4 fluxes from the arable plots significantly exceeded CH_4 fluxes from the grassland plots. However, all measured CH_4 fluxes were very low and CH_4 emissions and uptakes were almost negligible in the NECB of the plots, as was also reported by several other authors for drained organic soils (e.g. Maljanen et al., 2010; Petersen et al., 2012; Schäfer et al., 2012; Drösler et al., 2013; Renou-Wilson et al., 2014). Moreover, the C import through fertilization contributed only marginally (3–14 %) to the NECB of the grassland plots.

In the course of the present study, fertilization was found to enhance N_2O fluxes at the grassland sites, where the application of biogas digestate led to significantly higher N_2O emissions compared to cattle slurry application (for further discussion see Eickenscheidt et al., 2014b). From a meta-study of European organic soils Leppelt et al. (2014) found that the amount of N fertilizer was directly linked to N_2O fluxes from grasslands, whereas no significant relationship between N fertilization and N_2O fluxes from arable lands were found. Nevertheless, N_2O fluxes from the arable plots significantly exceeded those of the grassland sites, as was also reported by Maljanen et al. (2007, 2010) and Petersen et al. (2012) and additionally confirmed by Leppelt et al. (2014) for European organic soils. Observed N_2O peaks at the arable sites can be related to harvesting and/or several consecutive tillage steps (e.g. ploughing, milling, mattocking) in the previous weeks. This is in line with Silvan et al. (2005) who supposed that higher N_2O fluxes from arable lands are related to the higher N availability for microbial denitrification in the absence of plants. No fertilizer was applied at the arable plots, which is also reflected in the significantly lower N_{min} contents and perhaps higher pH values compared to the grassland plots. However, it is well known that drainage and intensive management enhanced the degradation of SOM and thus stimulates net nitrogen mineralization and nitrogen transformation processes (Kasimir Klemedtsson et al., 1997; Freibauer et al., 2004; Goldberg et al., 2010). Several authors reported an annual N supply through peat mineralization of approximately 70–425 $kg\,N\,ha^{-1}\,yr^{-1}$ (Schothorst, 1977; Flessa et al., 1998; Sonneveld and Lantinga, 2011; Leppelt et al., 2014). Taking into account the calculated soil carbon losses and plot specific C/N ratios of the upper soil/peat layer, estimated SOM mineralization leads to an annual N supply of approximately 451–1720 $kg\,N\,ha^{-1}\,yr^{-1}$. This estimation seems very high but regardless of the high uncertainties it clearly indicates that sufficient N must be

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available for nitrification and denitrification, independent of fertilizer application as previously assumed by Leppelt et al. (2014). Furthermore, the admixture of *Vicia sativa* or *Vicia faba minor*, both N_2 fixing leguminoses further increase the soil N_{min} pool of the arable sites through the release of N-rich root exudates (Rochette et al., 2004; Sey et al., 2010) as well as their incorporation into the soil, albeit to an unknown extent.

In conclusion, taking together estimated GHG emissions, calculated GWPs clearly differ between the two land-use types investigated with distinctly higher GWP's observed at the grassland plots compared to the arable land. However, all plots show a very high release of GHGs, demonstrating the unsustainable agricultural use of drained organic soils and the current need for the implementation of mitigation strategies and restoration measures. We hypothesized that GHG emissions from arable soils exceed GHG emissions from intensively managed grassland soils. The contrary was found in the present study; therefore we have to reject hypothesis ii. However, from the present results it can be concluded that mainly the management and not the land-use type itself or the SOC content is responsible for the amount of released GHGs from intensive farming on drained organic soils.

4.5 Implications for the climate reporting under LULUCF/AFOLU

For the climate reporting under LULUCF/AFOLU, the IPCC guidelines consider GHG emissions from peat soils having at least ≥ 10 cm thickness of the soil/peat layer and a C_{org} content of $\geq 12\,\%$ in case of a soil thickness ≤ 20 cm. However, the intensive cultivation of organic soils leads to a continuous decrease in the amount of SOM and thus the area of soils which fulfil the requirements of the IPCC guidelines for organic soils rapidly declined in the last decades. For example Nielsen et al. (2012) reported an average annual decrease of organic soils of approximately 1400 ha in Denmark since 1975. The remaining soils often contain $> 6\,\%$ C_{org} and not the required $> 12\,\%$ (Nielsen et al., 2012). Contrary to mineral soils or natural peatlands in equilibrium, Nielsen et al. (2012) assume that drained and managed soils having $> 6\,\%$ C_{org} will evidently lose carbon until a new equilibrium is reached. Since no data was available in

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literature for those soils, Nielsen et al. (2012) decided to allocate an fixed emission factor half of the amount of what was measured for soils having > 12 % C_{org} to account for these losses in the Danish greenhouse gas inventory. However, despite being subject to high uncertainties, our results reveal that the GHG emission potential of soils intermediate between mineral and organic soils can be as high or partly higher as for typical drained organic soils under intensive agricultural use. This is in line with observations from Leiber-Sauheitl et al. (2013) for extensive grasslands. To avoid a significant underestimation of GHG emissions in the LULUCF/AFOLU sector, there is a corresponding need to adjust the IPCC guidelines for drained inland organic soils accordingly. The new 2013 Supplement to the IPCC guidelines for national GHG inventories on wetlands distinguishes several emission factors for different land-use types, climate regions, nutrient status and drainage intensities (IPCC, 2014). We suggest establishing a further category which provides emission factors for different land-use types on former drained peatlands or associated organic soils, which do not fulfil the necessary requirements of typical organic soils but also contain high amounts of C_{org}. To define reliable emissions factors for those soils further investigations regarding their potential to release GHGs are needed. Furthermore, it has to be clarified to what extent the composition of the SOM is responsible for the magnitude of GHG release from drained organic soils.

5 Conclusions

This study presents estimations of GHG fluxes from arable lands and intensive grasslands on sapric Histosol and mollic Gleysol, which significantly differed in their SOC content in the top soil. The present results clearly revealed that like typical drained peatlands also drained mollic Gleysols can be considered as hotspots for GHG emissions, provided that they are intensively managed as arable land or grassland. However, observed GHG fluxes revealed a very high sensitivity against changing key factors like climate variables (e.g. temperature, precipitation) and management. Estimated GHG emission factors partly more than doubled the emission factor of the Tier 1 approach

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of the IPCC independent of the SOC content in the topsoil. Thus former peatlands and associated organic soils, containing < 12 % C_{org} should be integrated in the national GHG emission inventories to avoid a significant underestimation in the climate reporting. Moreover there is a current need to adjust the IPCC guidelines for drained inland organic soils accordingly. Besides climate reporting, the observed very high release of GHGs demonstrates the unsustainable agricultural use of drained organic soils and the current need for rapid implementation of mitigation strategies and restoration measures.

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Table 1. Physical and chemical properties of the investigated plots.

Site	Soil type	Organic carbon [%]		C/N ratio	pH ($CaCl_2$)	Bulk density [$g\,cm^{-3}$]		Mean GW level above surface [cm]	
		0–10 cm	10–20 cm	0–20 cm	0–20 cm	0–10 cm	10–20 cm	2010	2011
$A1C_{medium}$	molic Gleysol	9.6 ± 0.1	9.3 ± 0.2	10	5.24	0.72 ± 0.03	0.85 ± 0.08	−56 (−86/0)	−67 (−86/−4)
$A1C_{high}$	sapric Histosol	16.9 ± 0.2	17.2 ± 0.2	12	5.61	0.63 ± 0.05	0.67 ± 0.04	−45 (−90/7)	−49 (−76/5)
$A2C_{medium}$	molic Gleysol	9.4 ± 0.	9.2 ± 0.1	10	5.24	0.83 ± 0.06	0.90 ± 0.06	−56 (−86/0)	−67 (−86/−4)
$A2C_{high}$	sapric Histosol	16.1 ± 0.9	16.8 ± 0.2	12	5.61	0.67 ± 0.11	0.77 ± 0.08	−45 (−90/7)	−49 (−76/5)
$G1C_{medium}$	molic Gleysol	10.5 ± 0.2	9.4 ± 0.1	10	4.10	0.71 ± 0.09	0.90 ± 0.06	−65 (−91/−2)	−72 (−92/0)
$G1C_{high}$	sapric Histosol	17.2 ± 0.	16.7 ± 0.1	11	4.24	0.53 ± 0.09	0.64 ± 0.05	−45 (−64/−1)	−52 (−66/−3)
$G2C_{medium}$	molic Gleysol	10.9 ± 0.2	10.1 ± 0.1	10	4.10	0.81 ± 0.09	0.88 ± 0.03	−63 (−92/0)	−72 (−97/0)
$G2C_{high}$	sapric Histosol	16.4 ± 0.1	15.6 ± 0.1	11	4.24	0.57 ± 0.08	0.67 ± 0.03	−45 (−67/−1)	−50 (−65/−3)

Values present means ± SD.
Values in brackets are minimum and maximum values.
A, arable land.
G, grassland.

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Table 2. Date and type of conducted management events.

Date	Julian day	Management events A1	A2	G
24 Sep 2009	–	seed sowing (*Secale cereale*)	seed sowing (*Secale cereale*)	–
26 Mar 2010	85	–	–	levelling
30 Mar 2010	89	–	plowing and seed sowing (*Avena sativa* + 15% *Vicia faba minor*)	–
7 Apr 2010	97	–	–	rolling
13 Apr 2010	103	–	harrowing	–
28 Apr 2010	118	plowing	–	–
30 Apr 2010	120	seed sowing (*Zea mays*)	–	–
24 May 2010	144	grubbering	–	harvesting
11 Jun 2010	162	grubbering	–	–
14 Jun 2010	165	–	–	manuring [$20\,m^3\,ha^{-1}$]
6 Jul 2010	187	grubbering and hilling	–	–
20 Aug 2010	232	–	–	harvesting
22 Aug 2010	234	–	harvesting	–
25 Aug 2010	237	–	–	manuring [$20\,m^3\,ha^{-1}$]
28 Aug 2010	240	–	milling	–
4 Sep 2010	247	–	–	–
23 Sep 2010	266	–	–	herbicide against common sorrel (*Rumex acetosa*)
15 Oct 2010	288	harvesting	–	–
30 Oct 2010	303	mulching	–	–
16 Mar 2011	440	–	–	levelling
1 Apr 2011	456	plowing and seed sowing (*Avena sativa* + 20 % *Vicia sativa*)	–	–
18 Apr 2011	473	–	plowing	–
26 Apr 2011	481	–	grubbering + seed sowing (*Zea mays*)	–
30 Apr 2011	485	harrowing	harrowing	–
8 May 2011	493	–	harrowing	–
19 May 2011	504	–	mattocks	–
23 May 2011	508	–	–	harvesting
27 May 2011	512	–	–	manuring [$25\,m^3\,ha^{-1}$]
14 Jun 2011	530	–	hilling	–
1 Aug 2011	578	–	–	harvesting
16 Aug 2011	593	harvesting	–	–
18 Aug 2011	595	milling	–	–
27 Aug 2011	604	plowing and seed sowing (*Secale cereale*)	–	–
13 Sep 2011	621	–	–	harvesting
22 Sep 2011	630	–	–	manuring [$20\,m^3\,ha^{-1}$]
28 Sep 2011	636	–	harvesting	–

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Table 3. Cumulative R_{ECO}, GPP, NEE, CH_4 and N_2O exchange rates as well as C import through fertilizer and C export due to crop/grass yield.

Plot/year	R_{ECO} [$g C m^{-2} yr^{-1}$]	GPP [$g C m^{-2} yr^{-1}$]	NEE [$g C m^{-2} yr^{-1}$]	Fertilizer input* [$g C m^{-2} yr^{-1}$]	Yield* [$g C m^{-2} yr^{-1}$]	CH_4* [$g C m^{-2} yr^{-1}$]	N_2O* [$g N m^{-2} yr^{-1}$]
$A1C_{medium}/10$	2473 ± 272	−1454 ± 114	1019 ± 386	–	193 ± 53	–	–
$A1C_{medium}/11$	2992 ± 230	−1862 ± 126	1130 ± 356	–	74 ± 8	0.51 ± 0.17	0.27 ± 0.01
$A1C_{high}/10$	2012 ± 284	−873 ± 110	1139 ± 394	–	58 ± 23	–	–
$A1C_{high}/11$	2117 ± 123	−1302 ± 77	815 ± 200	–	135 ± 7	0.22 ± 0.04	0.23 ± 0.05
$A2C_{medium}/10$	2704 ± 544	−1449 ± 103	1255 ± 647	–	227 ± 27	–	–
$A2C_{medium}/11$	2354 ± 309	−2360 ± 237	−6 ± 546	–	457 ± 71	−0.03 ± 0.05	0.39 ± 0.06
$A2C_{high}/10$	2907 ± 482	−1200 ± 137	1707 ± 619	–	145 ± 19	–	–
$A2C_{high}/11$	2538 ± 329	−2188 ± 253	350 ± 582	–	330 ± 79	−0.10 ± 0.07	0.86 ± 0.21
$G1C_{medium}/10$	3954 ± 671	−2131 ± 180	1823 ± 851	126	297 ± 32	–	–
$G1C_{medium}/11$	4099 ± 300	−2414 ± 195	1685 ± 495	267	344 ± 63	−0.06 ± 0.09	0.12 ± 0.01
$G1C_{high}/10$	3736 ± 491	−2152 ± 140	1584 ± 631	126	325 ± 41	–	–
$G1C_{high}/11$	4026 ± 707	−2633 ± 138	1393 ± 845	267	455 ± 41	−0.07 ± 0.02	0.18 ± 0.02
$G2C_{medium}/10$	3683 ± 453	−2131 ± 213	1552 ± 666	76	342 ± 39	–	–
$G2C_{medium}/11$	4265 ± 379	−2880 ± 177	1385 ± 556	53	543 ± 58	−0.11 ± 0.05	0.19 ± 0.02
$G2C_{high}/10$	3521 ± 1041	−2093 ± 152	1428 ± 1193	76	380 ± 43	–	–
$G2C_{high}/11$	4316 ± 562	−2962 ± 178	1354 ± 740	53	593 ± 132	−0.02 ± 0.02	0.31 ± 0.09

* Data from grassland plots derived from Eickenscheidt et al. (2014).
A, arable land.
G, grassland.
10, year 2010.
11, year 2011.

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Table 4. Model evaluation statistics from observed R_{ECO} vs. modeled R_{ECO}.

Site	2010 r	NSE	PBIAS	RSR	2011 r	NSE	PBIAS	RSR
$A1C_{medium}$	0.90	0.70	−7.93	0.55	0.98	0.95	−0.17	0.22
$A1C_{high}$	0.98	0.96	0.44	0.19	0.98	0.97	1.79	0.18
$A2C_{medium}$	0.93	0.81	−5.68	0.44	0.94	0.89	−0.23	0.33
$A2C_{high}$	0.96	0.92	2.60	0.29	0.98	0.96	0.00	0.20
$G1C_{medium}$	0.96	0.93	1.54	0.27	0.95	0.91	−2.40	0.31
$G1C_{high}$	0.89	0.75	−6.27	0.50	0.97	0.95	0.03	0.23
$G2C_{medium}$	0.93	0.86	0.80	0.37	0.98	0.96	0.06	0.19
$G2C_{high}$	0.93	0.82	−4.65	0.42	0.97	0.94	0.92	0.25

r = Pearson's correlation coefficient.
NSE = Nash–Sutcliffe efficiency.
PBIAS = percent bias.
RSR = ratio of the root mean square error to the SD of measured data.

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Table 5. Model evaluation statistics from observed NEE vs. modeled NEE.

Site	2010 *r*	NSE	PBIAS	RSR	2011 *r*	NSE	PBIAS	RSR
$A1C_{medium}$	0.94	0.87	−11.84	0.36	0.97	0.93	1.41	0.26
$A1C_{high}$	0.94	0.88	−7.94	0.35	0.98	0.96	−4.94	0.21
$A2C_{medium}$	0.85	0.72	3.03	0.53	0.96	0.92	−3.64	0.28
$A2C_{high}$	0.79	0.61	3.63	0.63	0.96	0.91	−9.56	0.29
$G1C_{medium}$	0.90	0.80	−10.98	0.45	0.92	0.84	−10.47	0.40
$G1C_{high}$	0.91	0.82	−12.07	0.43	0.94	0.88	−10.04	0.35
$G2C_{medium}$	0.95	0.89	−13.23	0.33	0.96	0.92	−5.43	0.28
$G2C_{high}$	0.94	0.87	−10.71	0.36	0.94	0.89	−6.22	0.34

r = Pearson's correlation coefficient.
NSE = Nash–Sutcliffe efficiency.
PBIAS = percent bias.
RSR = ratio of the root mean square error to the SD of measured data.

Table 6. Estimated global warming potential for a time horizon of 100 years.

Site/periode	GWP_{100} $NEE_{corrected}$* [$g\,CO_{2\,eq.}\,m^{-2}\,yr^{-1}$]	GWP_{100} CH_4 [$g\,CO_{2\,eq.}\,m^{-2}\,yr^{-1}$]	GWP_{100} N_2O [$g\,CO_{2\,eq.}\,m^{-2}\,yr^{-1}$]	GWP_{100} balance [$g\,CO_{2\,eq.}\,m^{-2}\,yr^{-1}$]
$A1C_{medium}/11$	4419 ± 1336	16.96 ± 5.65	126.32 ± 4.68	4562 ± 1346
$A1C_{high}/11$	3487 ± 760	7.32 ± 1.33	107.61 ± 23.39	3601 ± 785
$A2C_{medium}/11$	1655 ± 2264	−1.00 ± 1.33	182.47 ± 28.07	1837 ± 2293
$A2C_{high}/11$	2496 ± 2426	−3.33 ± 1.66	402.36 ± 98.25	2895 ± 2526
$G1C_{medium}/11$	6467 ± 2048	−2.00 ± 2.99	56.14 ± 4.68	6521 ± 2056
$G1C_{high}/11$	5802 ± 3252	−2.33 ± 0.67	84.21 ± 9.36	5884 ± 3262
$G2C_{medium}/11$	6881 ± 2253	−3.66 ± 1.66	88.89 ± 9.36	6967 ± 2264
$G2C_{high}/11$	6951 ± 3200	−0.67 ± 0.67	145.04 ± 42.11	7095 ± 3243

* Corrected for C export and C import.

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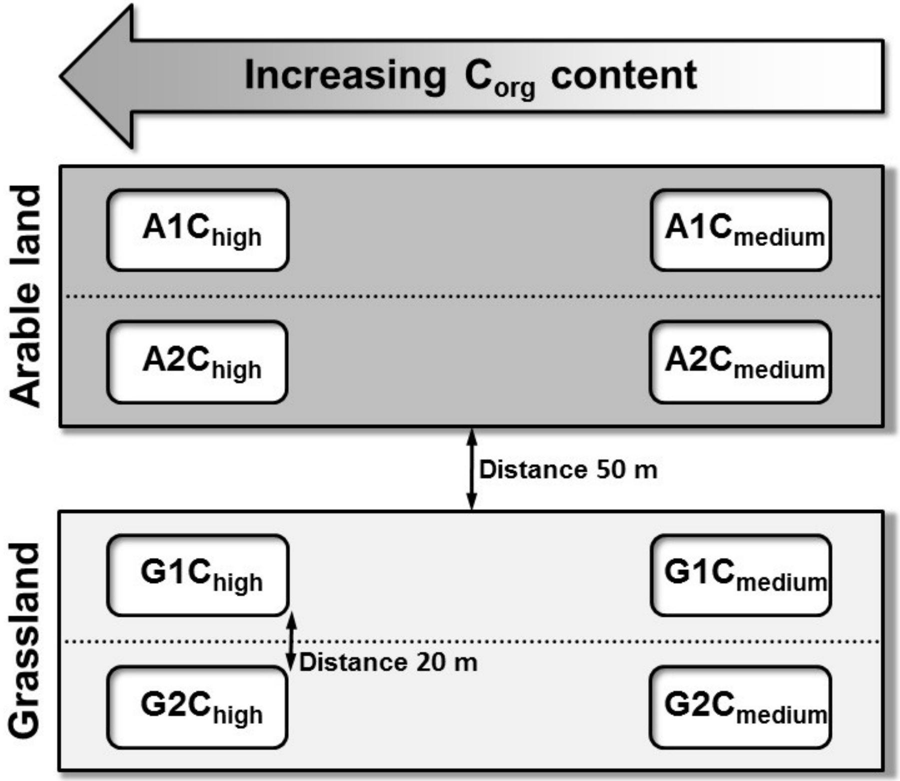

Figure 1. Schema of the experimental design.

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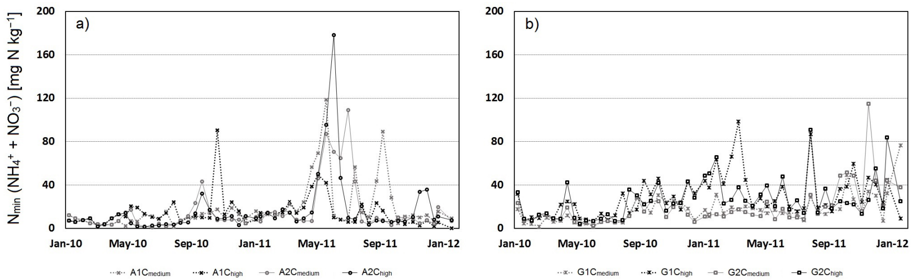

Figure 2. Mineral nitrogen contents [mg N kg^{-1}] for the arable land **(a)** and the grassland **(b)** of the soil depth 0–10 cm for the years 2010 and 2011.

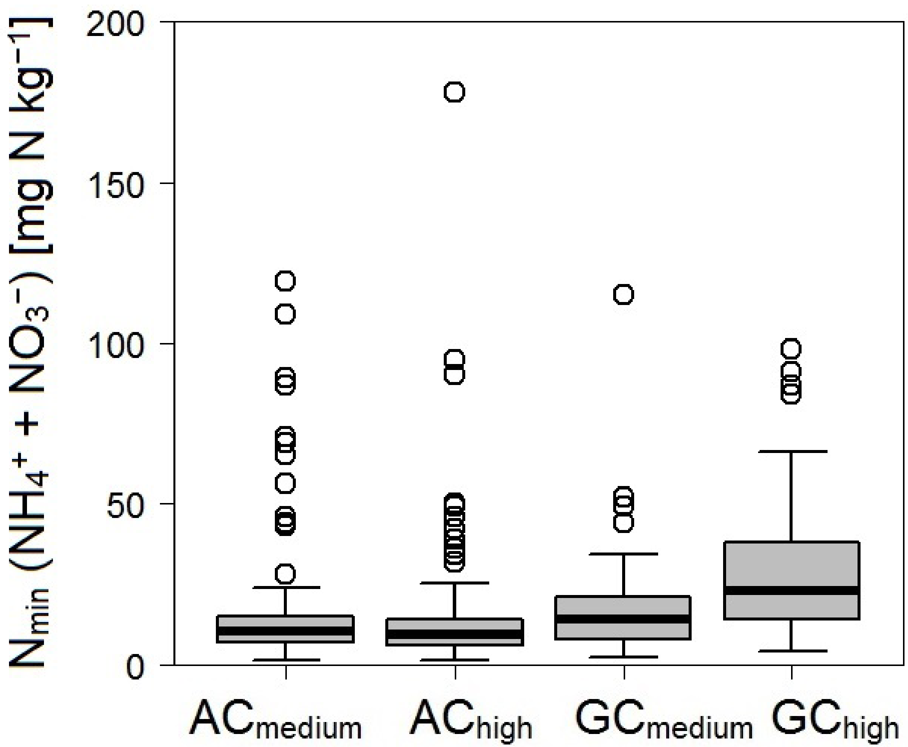

Figure 3. Box plots of mineral nitrogen contents [mg N kg^{-1}] of the soil depth 0–10 cm (A = arable land, G = grassland). Box plot showing median (central thick lines), 25 and 75 % quartile ranges around the median (box width). Circle present extreme values (≤ 1.5 times the interquartile range).

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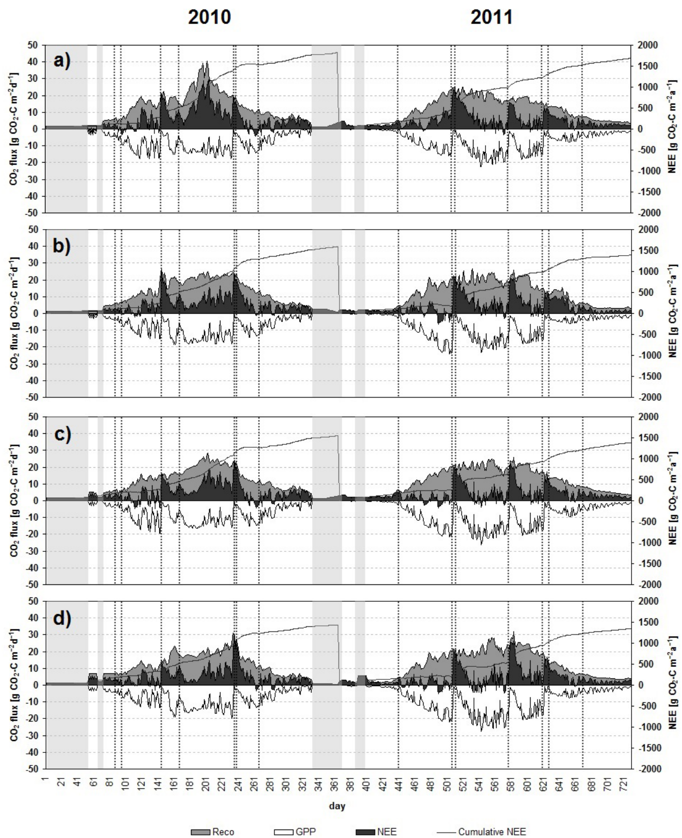

Figure 4. Time series of modeled CO_2 fluxes [$g\,CO_2\text{–}C\,m^{-2}\,d^{-1}$] and cumulative NEE [$g\,CO_2\text{–}C\,m^{-2}\,yr^{-1}$] for each site in 2010 and 2011; **(a)** grassland, cattle slurry, C_{medium}; **(b)** grassland cattle slurry, C_{high}; **(c)** grassland biogas digestate C_{medium}; **(d)** grassland, biogas digestate, C_{high}. Grey bars mark the period with snow cover. Dashed lines indicate management activities (see Table 2).

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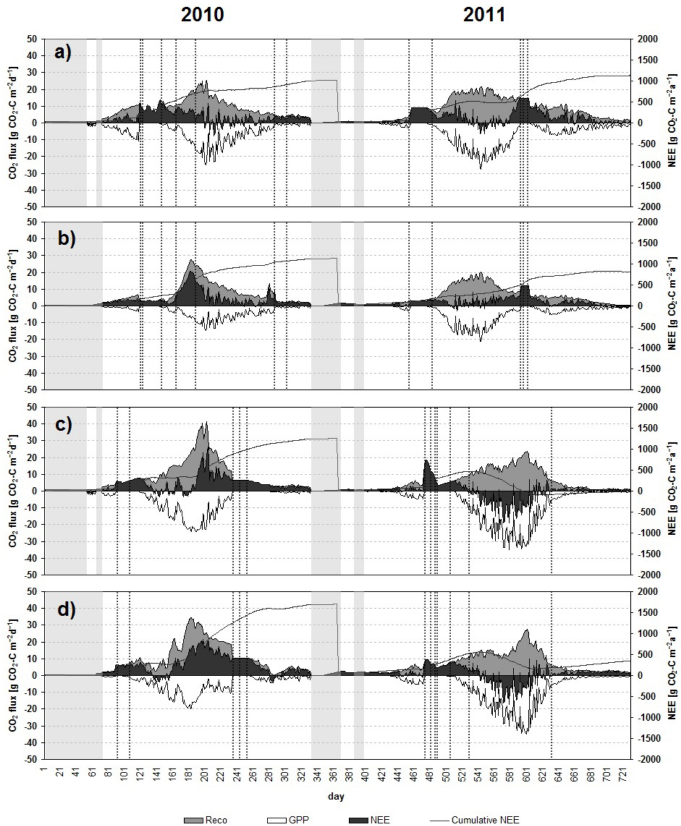

Figure 5. Time series of modeled CO_2 fluxes [$g\,CO_2$–$C\,m^{-2}\,d^{-1}$] and cumulative NEE [$g\,CO_2$–$C\,m^{-2}\,yr^{-1}$] for each site in 2010 and 2011; **(a)** arable land, 2010 maize, 2011 oat, C_{medium}; **(b)** arable land, 2010 maize, 2011 oat, C_{high}; **(c)** arable land, 2010 oat, 2011 maize, C_{medium}; **(d)** arable land, 2010 oat, 2011 maize, C_{high}. Grey bars mark the period with snow cover. Dashed lines indicate management activities (see Table 2).

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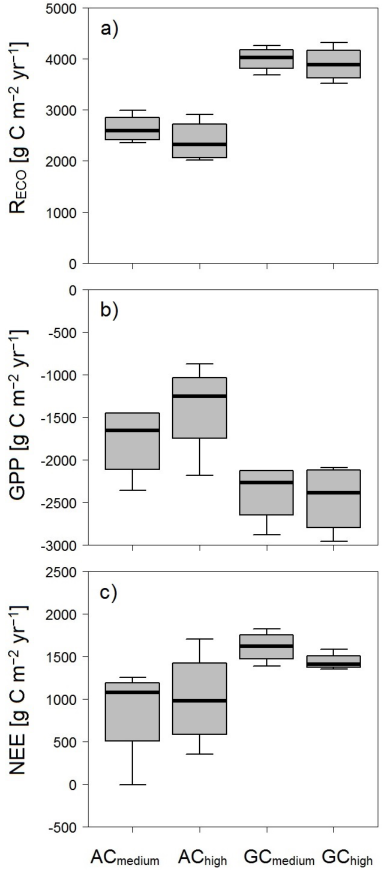

Figure 6. Box plots of cumulative R_{ECO} **(a)**, GPP **(b)** and NEE **(c)** for the two soil types and land-use types. Box plot showing median (central thick lines), 25 and 75 % quartile ranges around the median (box width).

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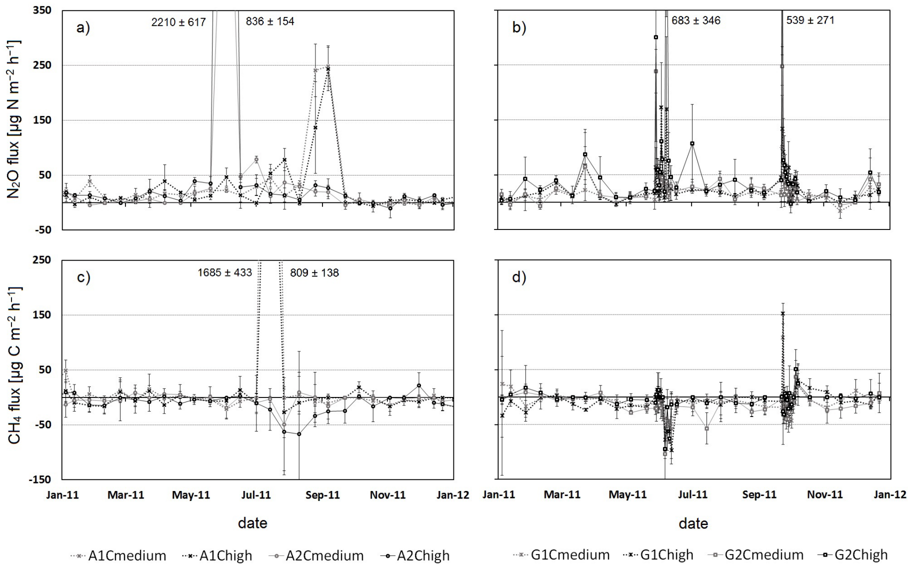

Figure 7. Time series of measured N_2O fluxes (**a**, arable land; **b**, grassland) and CH_4 fluxes (**c**, arable land; **d**, grassland) for the year 2011.

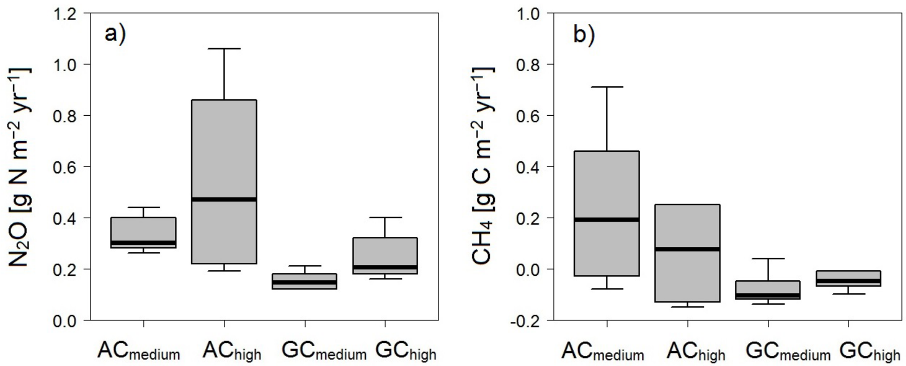

Figure 8. Box plots of cumulative annual N_2O emissions **(a)**, and cumulative annual CH_4 emissions for the two soil types and land-use types. Box plot showing median (central thick lines), 25 and 75 % quartile ranges around the median (box width).

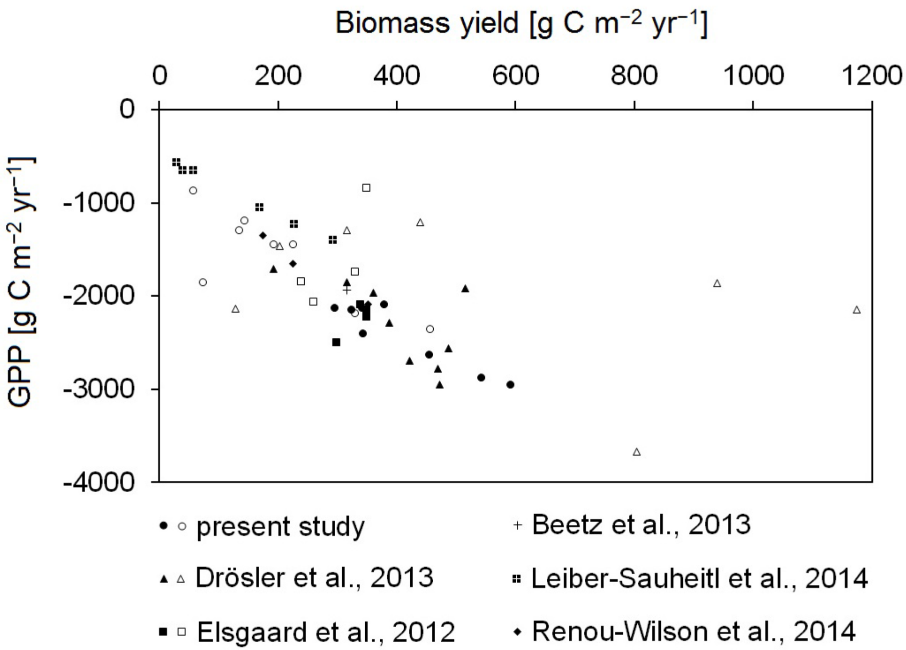

Figure 9. Relationship of GPP and biomass export from temperate peatlands. Filled symbols represents grassland sites (intensive and extensive), unfilled symbols represents arable lands.

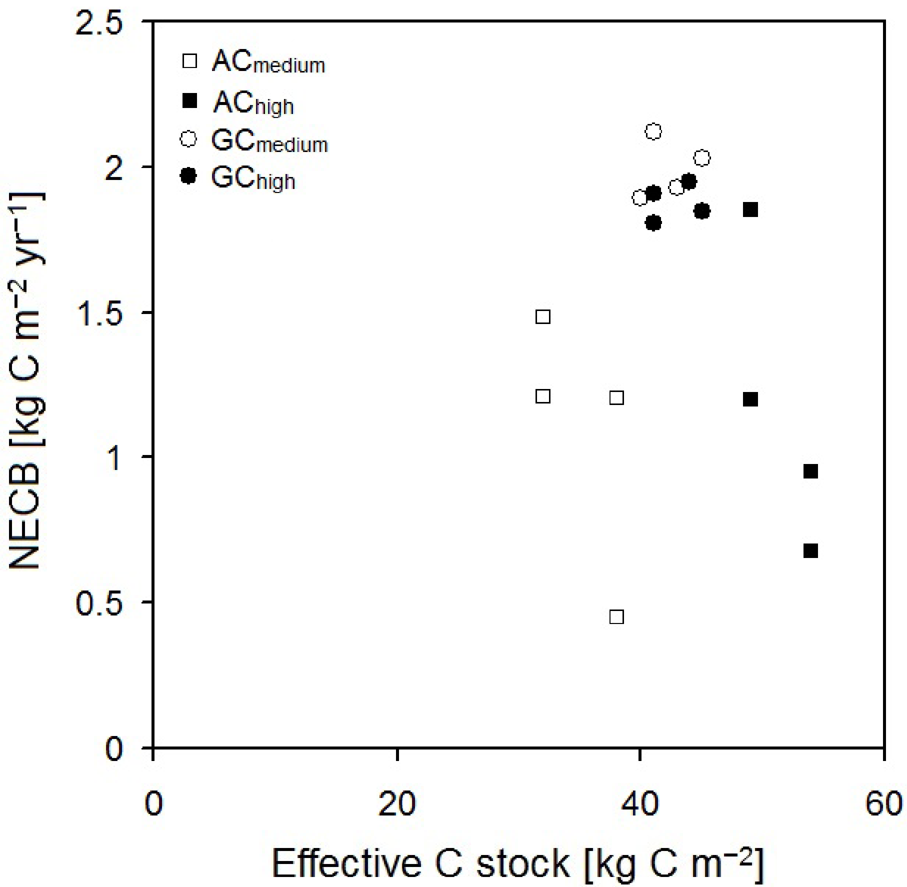

Figure 10. NECB plotted against the effective C stock, which is defined as the fraction of aerated carbon in the soil profile (according to Leiber-Sauheitl et al., 2014) (calculated NECB did not include CH_4 losses).

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