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 over a period of one or two years in case of N_2O/methane (CH_4) and CO_2, respectively. The daily and annual net ecosystem exchange (NEE) of CO_2 was determined by measuring NEE and the ecosystem respiration (R_{ECO}) with the closed dynamic chamber technique and by modeling the R_{ECO} and the gross primary production (GPP). N_2O and CH_4 were measured by the static 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 uncertainties, 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

Most 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 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 C_{org} to SOM 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 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 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 (1570 ha) is used as grassland and 20% as arable land (Schober et al., 2008).

According to the climate station at Munich airport, located 7 km east of the study sites, the 30-years mean annual temperature was 8.7 °C and the mean annual precipitation was 834 mm (1981–2010). 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 3) 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 (named C_{medium}) and 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) and Table 2. 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 x 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 set up at each site, centrally between the two plots (see Fig. 1; at the arable land, climate

stations represent temperatures from the management of the A1 plots) for the continuous recording (every 0.5 hour) 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 minutes. 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 Biomass yield, soil sampling and laboratory analyses

Crop and grass 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 3). To determine the annual crop/grass yield, samples were oven dried at 60°C for 48 hours 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 hours 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 cm³) 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 (0.78m x 0.78m x 0.5m; 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 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; sampling time was 0, 20, 40 and 60 minutes or 0, 40, 80, 120 minutes 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 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 and 2014b). The mentioned N_2O and CH_4 fluxes as well as soil properties, N_{min} values and biomass yield data from the grassland sites are derived from Eickenscheidt et al. (2014b).

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). Chamber 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 temperatures. 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 in the Supplement). 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, contrary to chambers used for N_2O/CH_4 measurements, three fans (SUNON® Super Silence MAGLev®-Lüfter) continuously operated during the CO_2 measurement to ensure a constant mixing of the chamber air (wind speed in chamber headspace ~1.5–2 m s^{-1}). 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.

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).

$$NEE = GPP + R_{ECO} \quad (1)$$

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).

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

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_0	activation energy [K]
T_{ref}	reference temperature: 283.15 [K]
T_0	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 the best explanatory power for R_{ECO}. At the grassland, we used site-specific climate station temperatures since we assume that they were comparable to plot-specific temperatures due to the comparable management and close proximity. At the A1 plots, R_{ECO} modeling based on plot-specific climate station

temperature files, whereas at the A2 plots, R_{ECO} modeling based likewise on the continuous climate data set of the A1 plots. This procedure probably produced some uncertainty for R_{ECO} modelling at the A2 plots, but due to the inaccuracy in manually observed temperatures, plot-specific temperature model building would have resulted in a higher uncertainty at these two plots. In case that the temperature span was too small for model building (e.g. winter time, snow cover) or a significant relationship between R_{ECO} and the temperature could not be observed (e.g. after ploughing), an average CO_2 flux was calculated for the measurement campaign. Annual 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 site or plot specific 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).

$$GPP = \frac{\alpha \cdot PAR}{\left(1 - \left(\frac{PAR}{2000}\right) + \left(\frac{\alpha \cdot PAR}{GPP_{2000}}\right)\right)} \quad (3)$$

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}]

Prior to modeling GPP, we corrected the plot specific PAR values since the acrylic glass of the transparent chambers reflects or 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, ploughing, 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 hour 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 Table S1 to S8 in the Supplement.

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, 1970), percent bias (PBIAS) and the ratio of the root mean square error to the standard deviation 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 3.0.1 (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 test (TukeyHSD). For GW level we used the non-parametric 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 2011, we used 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 non-parametric Mann-Whitney U-test. We accepted significant differences if $P \leq 0.05$. Results in the text are given as

means ± 1 standard error.

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. Annual mean air temperature in 20 cm height was 7.7 °C and 8.1°C at the GC_{medium} and GC_{high} sites in 2010 and 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. In 2010 annual mean air temperature in 20 cm height was 8.2°C and 8.1°C at the AC_{medium} and AC_{high} sites and 8.8°C and 8.7°C at the AC_{medium} and 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 1st of January to 12th of March 2010, 28th of November 2010 to 10th of January 2011 and from 24th of January to 5th of February 2011 (see also Fig. 4 and Fig. 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 1981–2010. 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 (Table1). 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 1st to 17th June 2010.

3.2 Soil properties and mineral nitrogen contents

Total organic carbon contents and bulk density in the 0–10 cm 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 10 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 (Fig. 2, Fig 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%) 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 or grass yield ranged from 58 ± 23 to 457 ± 71 g C m^{-2} yr^{-1} at the arable land and from 297 ± 32 to 593 ± 132 g C m^{-2} yr^{-1} at the grassland in 2010 and 2011 (see also Eickenscheidt et al., 2014b) (Table 4). For both land-use types the crop or grass yield was significatly ($P < 0.01$) lower in the year 2010 compared to the year 2011 (38% lower at the A sites and 31% lower 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 at both arable lands and used as silo maize or oat corn plus straw respectively, whereas in 2011 only the grains were harvested regarding both management practices and the remaining plants were left on the field (Table 3). 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 soil depth are the main drivers of R_{ECO} in the present study, while soil temperatures in −5 cm and −10 cm soil depth mostly showed distinctly weaker correlations (Table S1-S8 in the Supplement). 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} versus 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 observed R_{ECO} versus 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 5). 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-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-C m^{-2} d^{-1} and 32 g CO_2-C m^{-2} d^{-1} were modeled for the $A2C_{high}$ (maize) plot and the $G2C_{high}$ plot, respectively (Fig. 4 and 5). At the grassland sites, annual sums of modeled R_{ECO} ranged from 3521 ± 1041 ($G2C_{high}$/10) to 4316 ± 562 g CO_2-C m^{-2} yr^{-1} ($G2C_{high}$/11), which was significantly (P < 0.001) higher compared to the arable sites where R_{ECO} ranged from 2012 ± 284 ($A1C_{high}$/10, maize) to 2992 ± 230 g CO_2-C m^{-2} yr^{-1} ($A1C_{medium}$/11, oat; Table 4, 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-C m^{-2} d^{-1} were modeled for the arable land (maize, C_{medium}) and up to −20 g CO_2-C m^{-2} d^{-1} for the grassland ($G2C_{high}$), whereas in 2011, distinctly higher GPP values up to −35 g CO_2-C m^{-2} d^{-1} were modeled for both maize plots and up to −28 g CO_2-C m^{-2} d^{-1} for the $G2C_{high}$ plot (Fig. 4 and 5). At the grassland sites annual sums of GPP ranged between −2093 ± 152 ($G2C_{high}$/10) and −2962 ± 178 g CO_2-C m^{-2} yr^{-1} ($G2C_{high}$/11), which was significantly (P <0.01) higher compared to the arable sites where GPP ranged between −873 ± 110 ($A1C_{high}$/10, maize) and −2360 ± 237 g CO_2-C m^{-2} yr^{-1} ($A2C_{medium}$/11, maize; Table 4, 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 versus calculated NEE ranged from 0.79 to 0.98, NSE values ranged from 0.61 to 0.96 (Table 6). 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 −6 ± 546 ($A2C_{medium}$/11, maize) to 1707 ± 619 g CO_2-C m^{-2} yr^{-1} ($A2C_{high}$/10, oat), compared to the grassland sites were NEE ranged from 1354 ± 740 ($G2C_{high}$/11) to 1823 ± 851 g CO_2-C m^{-2} yr^{-1} ($G1C_{medium}$/10;Table 4, 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 µg N m^{-2} h^{-1}. However, single N_2O peaks with maximum flux rates of up to 2832 µg N m^{-2} h^{-1} were detected at the 3rd June at both maize plots as well as at the 6th of September at both oat plots with maximum flux rates of up to 289 µg N m^{-2} h^{-1}. At the grassland sites, highest N_2O fluxes of up to 992 µg 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 (Table 4). 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 4, 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 ± 0.01 g N m^{-2} yr^{-1} ($G1C_{medium}$) to 0.86 ± 0.21 g N m^{-2} yr^{-1} ($A2C_{high}$, maize; Table 4).

Most of the time, all sites showed a weak uptake of CH_4 or zero fluxes. CH_4 peaks up to 173 µg 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 µg C m^{-2} h^{-1} occurred on the 14th of 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 (Fig 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 ± 0.05 g C m^{-2} yr^{-1} ($G2C_{medium}$) and 0.51 ± 0.17 g C m^{-2} yr^{-1} ($A1C_{medium}$, oat; Table 4). However, as previously mentioned for N_2O, the single CH_4 peak event observed at the arable sites determine the cumulative sum of CH_4 and turns the plots from a sink into a source of CH_4.

3.6 NECB and GWP

Taking into consideration the C export from harvested phytomass, C import from fertilization, CH_4–C and CO_2–C exchange (NEE), calculated NECB ranged from 451 ± 617 ($A2C_{medium}$, maize) to 1894 ± 872 g C m^{-2} yr^{-1} ($G2C_{high}$). Estimated GWP's ranged from 1837 ± 2293 ($A2C_{medium}$, maize) to 7095 ± 3243 g $CO_{2eq.}$ m^{-2} yr^{-1} ($G2C_{high}$), revealing a very high release of greenhouse gases from all plots (Table 7). 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 or boreal drained arable lands (e.g. Maljanen et al., 2001 and 2007; Grønlund et al., 2008; Höper et al., 2008; Maljanen et al., 2010; Leifeld et al., 2011; Elsgaard et al., 2012; Drösler et al., 2013) and grasslands (e.g. Maljanen et al., 2001; Grønlund et al., 2006 and 2008; Maljanen et al., 2010; 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 0 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 et al., 1998; Flessa et al., 1997 and 1998; van Beek et al., 2010 and 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 & 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 used infrared gas analyzer LI-820 is just able to measure CO_2 concentrations, without consideration of spectral cross-sensitivity due to absorption band broadening and inherent instrument cross-sensitivity. Both cause an overestimation of CO_2 mole fraction in samples containing water vapour. Furthermore, the dilution effect of CO_2 in H_2O can cause a proportionate decrease in the sample CO_2 concentration. Particularly the increase of water vapour due to evaporation and/or transpiration leads to the fact that carbon uptake will be overestimated whereas the carbon release will respond vice versa (see Application Note #129 from LI-COR). This is in line with Pérez-Priego et al. (2015) who found that the increase of water vapour concentration in the headspace leads to one of the most important systematic errors affecting CO_2 flux estimations when using closed chambers provided that no corresponding correction is performed. According to Welles et al. (2001) the largest error due to increasing water vapour and the dilution effect will occur on wet soils with low CO_2 fluxes (dc/dt <1 ppm s^{-1}) and dry, sunny, conditions, when chamber air temperature and water vapour can rise rapidly. Only in advective high flux situations when the rate of increasing water vapour is less than 1% of the rate of increasing chamber CO_2, dilution effects may be ignored. This finding was also confirmed by Matsuura et al. (2011). However, neither corrections for cross-sensitivity and band broadening nor a dilution correction was applied in the present study. Nevertheless, the used cooling system partially reduced the dilution effect by ensuring a more or less constant air temperature and additionally by affecting air moisture and H_2O condensation, albeit to an unknown extent. However, it must be pointed out that modeled GPP will possibly be overestimated whereas modeled R_{ECO} will possibly be underestimated, resulting in significantly higher calculated NEE values. For future ecosystem CO_2-exchange studies we strongly recommend the use of a different infrared gas analyzer or the concurrently measurement of the relative humidity and temperature to perform a dilution correction to reduce significant errors in CO_2 flux measurements as proposed by Welles et al. (2001) and Pérez-Priego et al. (2015).

Secondly the R_{ECO} models 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 and Leiber-Sauheitl et al., 2014). Thirdly, some uncertainty in R_{ECO} models occurred

636 at both A2 plots since no plot specific temperature models were used. Due to the
637 inaccuracy of the manual determined temperatures we decided not to model plot-
638 specific temperatures for both A2 plots. However, we assume that the use of air
639 temperatures from climate stations of the adjacent arable plots is less problematic for
640 R_{ECO} modeling since 88% of R_{ECO} models were fitted to the air temperature which is
641 considered to be comparable between the two different plots. Fourthly, management
642 activities like ploughing at the arable sites probably produced peak CO_2 emissions,
643 which we may have missed. Additionally, it can be assumed that after harvesting at
644 the grassland sites, R_{ECO} decreased due to the reduced phytomass. However,
645 additional measurement campaigns to capture this effect did not take place in the
646 current study and no corresponding data were found in the literature. Furthermore, it
647 is well known that the application of organic fertilizers produced short-term CO_2
648 emission peaks, which were also not sufficiently detected. However, both sources of
649 errors may even have an opposite effect. Fifthly for GPP, linearly interpolation of
650 parameters produced some uncertainties since it can be assumed that plant growth
651 after cutting did not increase linearly (Horrocks and Valentine, 1999; Beetz et al.,
652 2013). However with the available data set, it was not possible to quantify the error
653 by the used interpolation approach of parameters since the dataset was too small for
654 cross validation and no additional measurements for an independent model
655 validation were conducted. In addition, despite high model accuracy, the calculated
656 PBIAS revealed a slightly model overestimation bias for R_{ECO} and NEE for both years
657 (R_{ECO} only in 2010). Thus, modeled R_{ECO} and calculated NEE rates should be
658 considered as a conservative estimation. However, modeled values fit well with
659 values reported in the literature (see Figure 9).
660 Several studies indicated that dissolved organic C can significantly contribute to
661 terrestrial C balances (e.g. Worrall et al., 2009; Dinsmore et al., 2010 Renou-Wilson
662 et al., 2014). Thus, for the calculation of NECB from drained organic soils, fluvial C
663 losses should additionally be considered in future investigations.
664 Observed N_2O fluxes showed a high temporal variability with long periods of low
665 background emissions and a few high peaks, mainly after management activities.
666 Measurement frequency was increased after fertilization at the grassland plots for at
667 least two weeks (see Eickenscheidt et al., 2014b) but due to our regular
668 measurement intervals in the remaining year we cannot rule out that we may have
669 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. Furthermore, Christiansen et al. (2011) and Juszczak (2013) found that fluxes estimated in non-mixed chambers (without fans) were significantly underestimated (up to 58%) compared to the measured reference fluxes. Moreover, all gas fluxes were calculated solely by ordinary linear regression models, which partially carries the risk to underestimate gas fluxes compared to non-linear functions (see e.g. Pihlatie et al., 2013). Thus it is possible that we systematically underestimated N_2O and CH_4 fluxes. However, for future investigations in GHG emissions we strongly advocate firstly the combined use of automatic and manual chamber systems and secondly the testing of linear versus non-linear models for gas flux calculation, to obtain a higher accuracy of data.

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). Also Pohl et al. (2015) found that the static SOC stocks showed no significant effects on C fluxes of maize in a heterogenous peatland, whereas the dynamic C

704 (SOC_{dyn}) and N (N_{dyn}) stocks and their interaction with GW level strongly influenced
705 the C gas exchange. We additionally tried to apply the concept of SOC_{dyn} and N_{dyn}
706 stocks as described in Pohl et al. (2015), but contrasting to them neither the GW
707 level nor the SOC_{dyn} or N_{dyn} had any explanatory power in our study. However, Fig.
708 10 shows that at the grassland sites, C stocks available for mineralization processes
709 are comparable (40–45 kg C m^{-2}), probably explaining the equal CO_2 loss rates from
710 this land-use type. Temperature and soil moisture are considered to be the primary
711 regulators for CO_2 emissions from soils (Silvola et al. 1996; Maljanen et al., 2001;
712 Hardie et al., 2011), since they directly affect microbial activity and the rate of
713 enzymatic processes (Michaelis and Menten, 1913; Tietema et al., 1992). In the
714 present study, temperatures are found to be equal at all sites due to their close
715 proximity, whereas the soil moisture contents significantly differed between the C_{high}
716 and C_{medium} sites mainly according to the GW oscillation. It is well known that the
717 water level height has a strong influence on CO_2 emissions from peatlands as it
718 directly affects the oxygen availability for microbial activity as was reported in several
719 studies (e.g. Silvola et al., 1996; Berglund and Berglund, 2011; Renou-Wilson et al.,
720 2014; Leiber-Sauheitl et al., 2014). Beside abiotic factors substrate chemistry, in
721 particular the SOM quality and its labile and more recalcitrant fractions, are
722 considered to act as key variables affecting the decomposability of SOM and thus
723 controlling CO_2 fluxes from peatlands (Byrne and Farrell, 2005; Heller and Zeitz,
724 2012; Leifeld et al., 2012). For example, Leifeld et al. (2012) showed that the soil
725 respiration rate of a disturbed temperate peatland was strongly controlled by its
726 polysaccharides content, particularly the O-alkyl-C content was found to be a useful
727 proxy for respiration rates. SOM quality was not examined in our study, but both soil
728 types at all plots investigated exhibited highly decomposed organic material (H10,
729 according to Von Post's humification scale; N. Roßkopf personal communication,
730 2013). This is typical for organic soils which have been drained and intensively
731 managed for a long time, and is in line with Leifeld et al. (2012), who found that
732 organic matter quality declines with ongoing decomposition, resulting in low
733 polysaccharides contents and a lower availability for heterotrophic metabolism.
734 Nevertheless, observed NECB revealed very high C loss rates from the SOC pool.
735 Leifeld et al. (2014) suggested that intensive management, drainage and changed
736 climate drivers accelerate peat decomposition today, and therefore outweighed
737 declining peat quality. Additionally, Reiche et al. (2010) reported that the degree of

738 humification is not suitable for the prediction of CO_2 and CH_4 fluxes from anaerobic
739 decomposition, which stands in contrast to assumptions made by Glatzel et al.
740 (2004). However, observed equal narrow C/N ratios (10–12) in the upper soil reveal
741 firstly a high organic matter quality, easily to mineralize, and secondly comparable
742 SOM qualities at all plots, probably explaining why no significantly different C loss
743 rates between the two different soil types were found in the present study.
744 In line with CO_2, CH_4 fluxes were also not different between the two soil types
745 investigated, but this can mainly be attributed to the intensive drainage and thus soil
746 aeration, which effectively inhibited microbial methanogenesis at the C_{medium} and C_{high}
747 sites. It is known that the availability and quality of organic substrates influences the
748 amount of produced CH_4. Nevertheless, several studies indicate that high CH_4 fluxes
749 in bogs are mainly controlled by labile organic substrates such as root exudates or
750 plant litter and not by bulk peat (Minchin & McNaughton, 1984; Chanton et al., 1995;
751 Bridgham et al., 1998; Whalen, 2005; Hahn-Schöfl et al., 2011).
752 In contrast to CO_2 and CH_4 fluxes, N_2O fluxes from the C_{high} sites significantly
753 exceeded N_2O fluxes from the C_{medium} sites. This can probably be attributed to the
754 more favorable soil conditions for denitrification, supported by higher N_{min} contents
755 and higher groundwater levels at these sites (Eickenscheidt et al., 2014b). In both
756 years N_{min} was mainly dominated by NO_3^-, demonstrating that net nitrification entirely
757 controls net nitrogen mineralization at all plots. Thus, nitrification provided the
758 substrate for denitrification and additionally, may itself have contributed to N_2O
759 production. In general, N_2O production processes are various and can occur
760 simultaneously within close proximity (Davidson et al., 1986; Butterbach-Bahl et al.,
761 2013). Both nitrification as well as denitrification depend on the availability of labile
762 organic compounds as C and/or energy source (Butterbach-Bahl et al., 2013), in
763 which autotrophic nitrification depends particularly on the availability of CO_2 for cell
764 growth (Delwiche and Finstein, 1965). However, for denitrification the actual
765 regulation by C is currently not yet understood (Baggs and Philippot, 2011), but it can
766 be assumed that sufficient metabolizable C was widely available at all plots
767 investigated.

768 ## 4.4 Land-use and management effects

769 At peatlands GW level and land-use type are closely linked. From a meta-analysis of
770 53 German peatlands Tiemeyer et al. (2013) found that the mean annual GW level
771 was lower for arable land than for intensive grassland with median GW levels of

772 approximately −70 and −37 cm below soil surface. The GW levels observed in our
773 study were on average lower at the arable land and higher at the grassland
774 compared with the average of the meta-analysis. In general, intensive farming at
775 peatlands presupposes low GW levels, since most of the arable crops are not
776 adapted to low oxygen contents in the rhizosphere as could be seen in the present
777 study, where the temporarily high GW level or flooding caused plant damage and
778 yield losses at the arable sites in 2010. The effect of reduced biomass productivity
779 due to high GW levels which inhibited photosynthesis by slowing the rate of gas
780 diffusion through the vegetation (Lohila, 2008) was also reported by Renou-Wilson et
781 al. (2014). Both annual sums of GPP as well as yields were in good agreement with
782 those reported from other peatlands as can be seen in Figure 9. Statistical analysis
783 revealed significantly higher yields at the grassland sites compared to the arable
784 sites, but it has to be taken into account that at the arable sites only the grains were
785 harvested in 2011 and up to 3.84 t DM ha^{-1} and 9.05 t DM ha^{-1} remained on the field
786 regarding the oat and maize plots, respectively. Due to the continuous plant cover
787 over the whole year at the grassland plots annual sums of GPP were significantly
788 higher at these plots compared to the arable plots in 2010 as well as in 2011.
789 As GPP, modeled annual sums of R_{ECO} significantly differed between the two land-
790 use types with distinctly higher R_{ECO} values at the grassland sites. As mentioned
791 above, R_{ECO} is strongly controlled by temperature since it stimulates both R_a and R_h,
792 as can be seen in the pronounced seasonality of R_{ECO}. From the model fits it can be
793 suggested that the more frequent model adaptation with T_{air} (88%) reveals a higher
794 share of R_a at the arable site compared to the grassland sites. At the later,
795 approximately 40% of the R_{ECO} models based on ST_2, perhaps demonstrating a more
796 balanced ratio of R_a and R_h. Nevertheless, the proportion of the different respiration
797 compartments of R_{ECO} is unknown, but Silvola et al. (1996) reported that root-derived
798 respiration from grasslands established on peatland accounted for 35–45% of total
799 soil respiration. Furthermore, Maljanen et al. (2001) found that root-associated
800 respiration on grasslands was distinctly higher compared to arable lands. However,
801 the significantly higher R_{ECO} at the grassland sites can firstly perhaps be related to
802 the higher biomass production at these sites, because a higher GPP also results in
803 higher above- and below-ground autotrophic respiration (Leiber-Sauheitl et al., 2014;
804 Renou-Wilson et al., 2014). Moreover, the increased transport of photosynthates to
805 the plant rhizosphere due to the higher GPP may favor bacterial metabolism through

806 increased root exudates (Mounier et al., 2004; Henry et al., 2008; Sey et al., 2010),
807 additionally enhancing R_h. Secondly, the organic fertilizer application at the grassland
808 plots stimulates microbial growth and thus SOM mineralization (Gutser et al., 2005;
809 Jones et al., 2007). Additionally, a large part of the C from the organic fertilizer will
810 quickly be metabolized to CO_2 (Vuichard et al., 2007). Several authors (see e.g. Dao,
811 1998; Maljanen et al., 2010) reported that regularly ploughed and fertilized arable
812 lands are larger sources of CO_2 than non-tilled arable land soils or grasslands, due to
813 aerating and mixing of crop residues into the soil. However, in the current study the
814 effect of management is difficult to capture.
815 Despite of higher modeled GPP values, the distinctly higher modeled R_{ECO} values led
816 to significantly higher calculated NEE values at the grassland sites compared to the
817 arable sites. With the exception of the maize plot at the C_{medium} site in the year 2011,
818 all plots show positive NEE balances in both years investigated, as expected for
819 drained organic soils and as commonly reported in the literature (e.g. Maljanen et al.,
820 2001; Grønlund et al., 2006 and 2008; Maljanen et al., 2010; Elsgaard et al., 2012;
821 Beetz et al., 2013; Drösler et al., 2013). However, the huge CO_2 uptake capacity
822 during the short growth period of the maize plants, compensates for the soil CO_2
823 release due to microbial decomposition of organic matter at least in the year 2011.
824 Nevertheless, as seen in the NECB, the C export also reversed the maize cultivation
825 on the C_{medium} site to a C source. Previous studies of annual NEE from maize on
826 organic soils are rare in literature, but our results are in line with Drösler et al. (2013)
827 who reported NEE values ranging from −216.2 to 443.8 g C m^{-2} yr^{-1}. As mentioned
828 above, it has to be taken into account that in the year 2011 only the grains were
829 harvested at all arable plots. Assuming that silage maize would have been produced
830 instead of maize grains or the straw was additionally harvested at the oat plots,
831 NECB would partly be doubled and more comparable to calculated grassland values.
832 According to Maljanen et al. (2010) the better aeration of regularly ploughed arable
833 land leads to a larger sink of atmospheric CH_4 compared to permanent grasslands.
834 This contrasted our results, where the CH_4 fluxes from the arable plots significantly
835 exceeded CH_4 fluxes from the grassland plots. However, all measured CH_4 fluxes
836 were very low and CH_4 emissions and uptakes were almost negligible in the NECB of
837 the plots, as was also reported by several other authors for drained organic soils (e.g.
838 Maljanen et al., 2010; Petersen et al., 2012; Schäfer et al., 2012; Drösler et al., 2013;
839 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 and 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 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 ≥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 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

908 land-use types on former drained peatlands or associated organic soils, which do not
909 fulfil the necessary requirements of typical organic soils but also contain high
910 amounts of C_{org}. To define reliable emissions factors for those soils further
911 investigations regarding their potential to release GHGs are needed. Furthermore, it
912 has to be clarified to what extent the composition of the SOM is responsible for the
913 magnitude of GHG release from drained organic soils.

914 5 Conclusion

915 This study presents estimations of GHG fluxes from arable lands and intensive
916 grasslands on sapric Histosol and mollic Gleysol, which significantly differed in their
917 SOC content in the top soil. Despite a high uncertainty in GHG flux estimations and
918 modeling, the present results clearly revealed that like typical drained peatlands also
919 drained mollic Gleysols can be considered as hotspots for GHG emissions, provided
920 that they are intensively managed as arable land or grassland. However, observed
921 GHG fluxes revealed a very high sensitivity against changing key factors like climate
922 variables (e.g. temperature, precipitation) and management. Estimated GHG
923 emission factors partly more than doubled the emission factor of the Tier 1 approach
924 of the IPCC independent of the SOC content in the topsoil. Thus former peatlands
925 and associated organic soils, containing <12% C_{org} should be integrated in the
926 national GHG emission inventories to avoid a significant underestimation in the
927 climate reporting. Moreover there is a current need to adjust the IPCC guidelines for
928 drained inland organic soils accordingly. Besides climate reporting, the observed very
929 high release of GHGs demonstrates the unsustainable agricultural use of drained
930 organic soils and the current need for rapid implementation of mitigation strategies
931 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 below 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 ± 0.0	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 ± 0.0	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;

Table 2 Physical and chemical properties of the applied slurries and digestates (data derived from Eickenscheidt et al., 2014b).

	Cattle slurry				Biogas digestate			
	1. Application (14.06.2010)	2. Application (25.08.2010)	3. Application (27.05.2011)	4. Application (22.09.2011)	1. Application (14.06.2010)	2. Application (25.08.2010)	3. Application (27.05.2011)	4. Application (22.09.2011)
Fertilizer quantity [m^3 ha^{-1}]	20	20	25	20	20	20	25	20
Total nitrogen [kg ha^{-1}]	47	64	70	85	49	52	78	35
NO_3^- [kg N ha^{-1}]	0	0	0	0	0	0	0	0
NH_4^+ [kg N ha^{-1}]	20	28	23	33	22	28	51	17
C/N ratio	12	11	11	9	8	7	2	5
pH ($CaCl_2$)	–	–	6.8	7.0	–	–	7.7	7.4

1437 **Table 3** Date and type of conducted management events.

Date	Julian day	Management events		
		A1	A2	G
2009-09-24	–	seed sowing (Secale cereale)	seed sowing (Secale cereale)	–
2010-03-26	85	–	–	levelling
2010-03-30	89	–	plowing & seed sowing (*Avena sativa* + *15*% *Vicia faba minor*)	–
2010-04-07	97	–	–	rolling
2010-04-13	103	–	harrowing	–
2010-04-28	118	plowing	–	–
2010-04-30	120	seed sowing (*Zea mays*)	–	–
2010-05-24	144	grubbering	–	harvesting
2010-06-11	162	grubbering	–	–
2010-06-14	165	–	–	manuring
2010-07-06	187	grubbering & hilling	–	–
2010-08-20	232	–	–	harvesting
2010-08-22	234	–	harvesting	–
2010-08-25	237	–	–	manuring
2010-08-28	240	–	milling	–
2010-09-04	247	–	–	–
2010-09-23	266	–	–	herbicide against common sorrel (*Rumex acetosa*)
2010-10-15	288	harvesting	–	–
2010-10-30	303	mulching	–	–
2011-03-16	440	–	–	levelling
2011-04-01	456	plowing & seed sowing (*Avena sativa* + 20% *Vicia sativa*)	–	–
2011-04-18	473	–	plowing	–
2011-04-26	481	–	grubbering + seed sowing (Zea mays)	–
2011-04-30	485	harrowing	harrowing	–
2011-05-08	493	–	harrowing	–
2011-05-19	504	–	mattocks	–
2011-05-23	508	–	–	harvesting
2011-05-27	512	–	–	manuring
2011-06-14	530	–	hilling	–
2011-08-01	578	–	–	harvesting
2011-08-16	593	harvesting	–	–
2011-08-18	595	milling	–	–
2011-08-27	604	plowing & seed sowing (*Secale cereale*)	–	–
2011-09-13	621	–	–	harvesting
2011-09-22	630	–	–	manuring
2011-09-28	636	–	harvesting	–

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Table 4 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	cultivated crop	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	silage maize	2473 ± 272	−1454 ± 114	1019 ± 386	–	193 ± 53	–	–
$A1C_{medium}$ / 11	oat grains	2992 ± 230	−1862 ± 126	1130 ± 356	–	74 ± 8	0.51 ± 0.17	0.27 ± 0.01
$A1C_{high}$ / 10	silage maize	2012 ± 284	−873 ± 110	1139 ± 394	–	58 ± 23	–	–
$A1C_{high}$ / 11	oat grains	2117 ± 123	−1302 ± 77	815 ± 200	–	135 ± 7	0.22 ± 0.04	0.23 ± 0.05
$A2C_{medium}$ / 10	oat grains + straw	2704 ± 544	−1449 ± 103	1255 ± 647	–	227 ± 27	–	–
$A2C_{medium}$ / 11	maize grains	2354 ± 309	−2360 ± 237	−6 ± 546	–	457 ± 71	−0.03 ± 0.05	0.39 ± 0.06
$A2C_{high}$ / 10	oat grains + straw	2907 ± 482	−1200 ± 137	1707 ± 619	–	145 ± 19	–	–
$A2C_{high}$ / 11	maize grains	2538 ± 329	−2188 ± 253	350 ± 582	–	330 ± 79	−0.10 ± 0.07	0.86 ± 0.21
$G1C_{medium}$ / 10	grass 2 cuts	3954 ± 671	−2131 ± 180	1823 ± 851	126	297 ± 32	–	–
$G1C_{medium}$ / 11	grass 3 cuts	4099 ± 300	−2414 ± 195	1685 ± 495	267	344 ± 63	−0.06 ± 0.09	0.12 ± 0.01
$G1C_{high}$ / 10	grass 2 cuts	3736 ± 491	−2152 ± 140	1584 ± 631	126	325 ± 41	–	–
$G1C_{high}$ / 11	grass 3 cuts	4026 ± 707	−2633 ± 138	1393 ± 845	267	455 ± 41	−0.07 ± 0.02	0.18 ± 0.02
$G2C_{medium}$ / 10	grass 2 cuts	3683 ± 453	−2131 ± 213	1552 ± 666	76	342 ± 39	–	–
$G2C_{medium}$ / 11	grass 3 cuts	4265 ± 379	−2880 ± 177	1385 ± 556	53	543 ± 58	−0.11 ± 0.05	0.19 ± 0.02
$G2C_{high}$ / 10	grass 2 cuts	3521 ± 1041	−2093 ± 152	1428 ± 1193	76	380 ± 43	–	–
$G2C_{high}$ / 11	grass 3 cuts	4316 ± 562	−2962 ± 178	1354 ± 740	53	593 ± 132	−0.02 ± 0.02	0.31 ± 0.09

Values present means ± SE
* Data from grassland plots derived from Eickenscheidt et al. (2014b).
A, arable land; G, grassland; 10, year 2010; 11, year 2011.

Table 5 Model evaluation statistics from observed R_{ECO} versus modeled R_{ECO}. *r* = Pearson's correlation coefficient, NSE = Nash-Sutcliffe efficiency, PBIAS = percent bias, RSR = ratio of the root mean square error to the standard deviation of measured data.

	2010				2011			
Site	*r*	NSE	PBIAS	RSR	*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

Table 6 Model evaluation statistics from observed NEE versus modeled NEE. *r* = Pearson's correlation coefficient, NSE = Nash-Sutcliffe efficiency, PBIAS = percent bias, RSR = ratio of the root mean square error to the standard deviation of measured data.

	2010				2011			
Site	*r*	NSE	PBIAS	RSR	*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

Table 7 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

Values present means ± SE
* Corrected for C export and C import

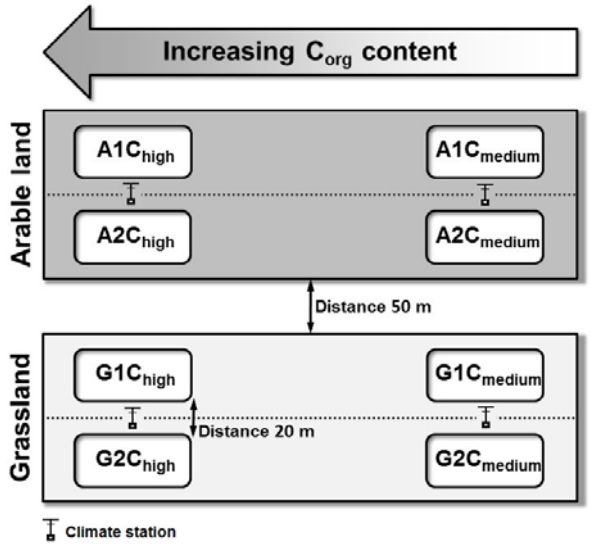


Fig. 1 Schema of the experimental design.

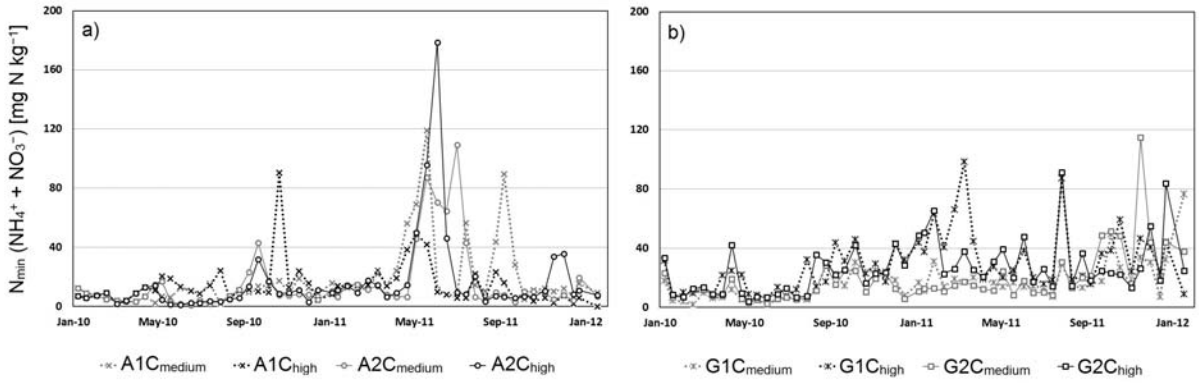


Fig. 2 Mineral nitrogen contents [mg N kg^{-1}] for the arable land a) and the grassland b) of the soil depth 0–10 cm for the years 2010 and 2011. Data from grassland plots (b) derived from Eickenscheidt et al. (2014b).

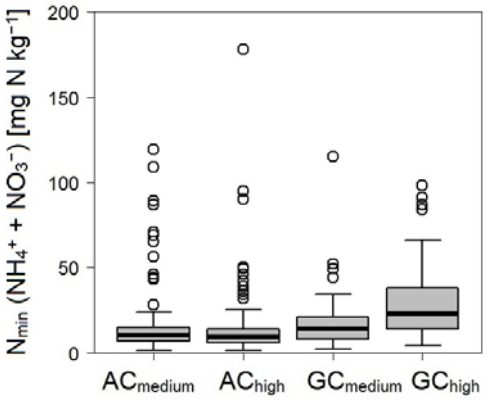
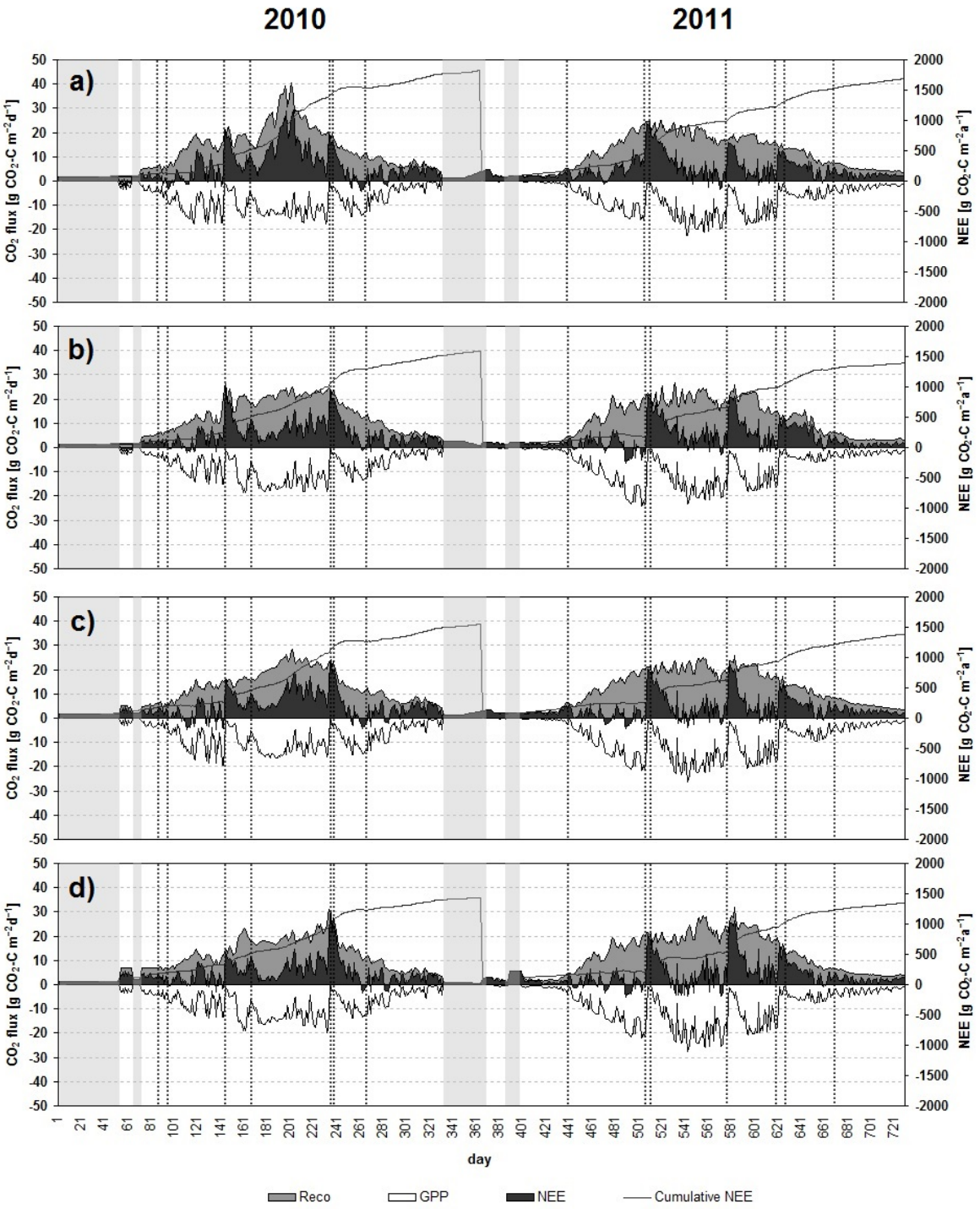


Fig. 3 Box plots of mineral nitrogen contents [mg N kg^{-1}] of the soil depth 0–10 cm (A = arable land, G = grassland). Box plot showing median (central thick lines), 25% and 75% quartile ranges around the median (box width). Circle present extreme values (≤1.5 times the interquartile range).



1495

1496 **Fig. 4** 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}]
1497 for each site in 2010 and 2011; a) grassland, cattle slurry, C_{medium}; b) grassland cattle slurry, C_{high}; c)
1498 grassland biogas digestate C_{medium}; d) grassland, biogas digestate, C_{high}. Grey bars mark the period
1499 with snow cover. Dashed lines indicate management activities (see Table 3).

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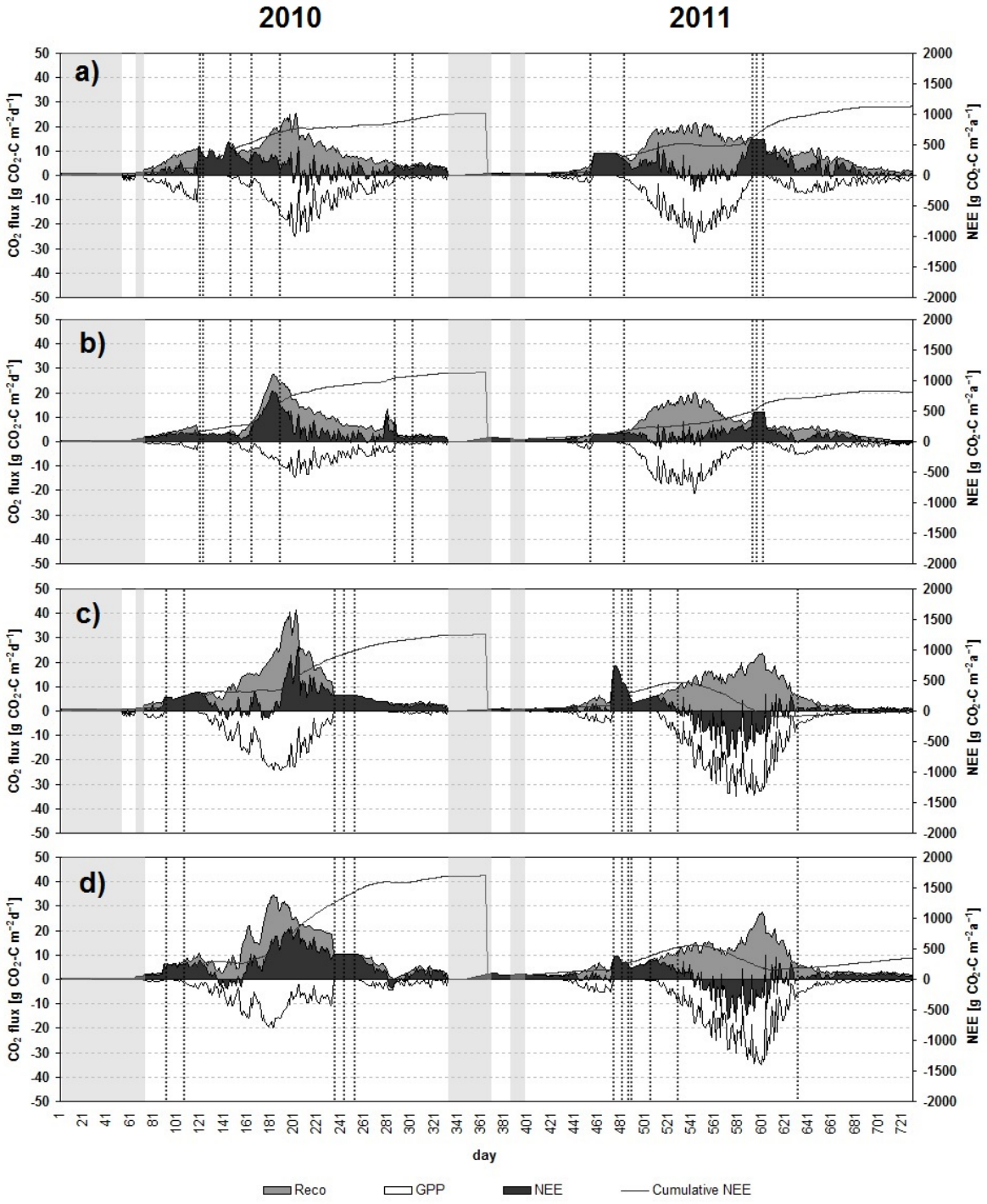


1502

1503 **Fig. 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}]
1504 for each site in 2010 and 2011; a) arable land, 2010 maize, 2011 oat, C_{medium}; b) arable land, 2010
1505 maize, 2011 oat, C_{high}; c) arable land, 2010 oat, 2011 maize, C_{medium}; d) arable land, 2010 oat, 2011
1506 maize, C_{high}. Grey bars mark the period with snow cover. Dashed lines indicate management activities
1507 (see Table 3).

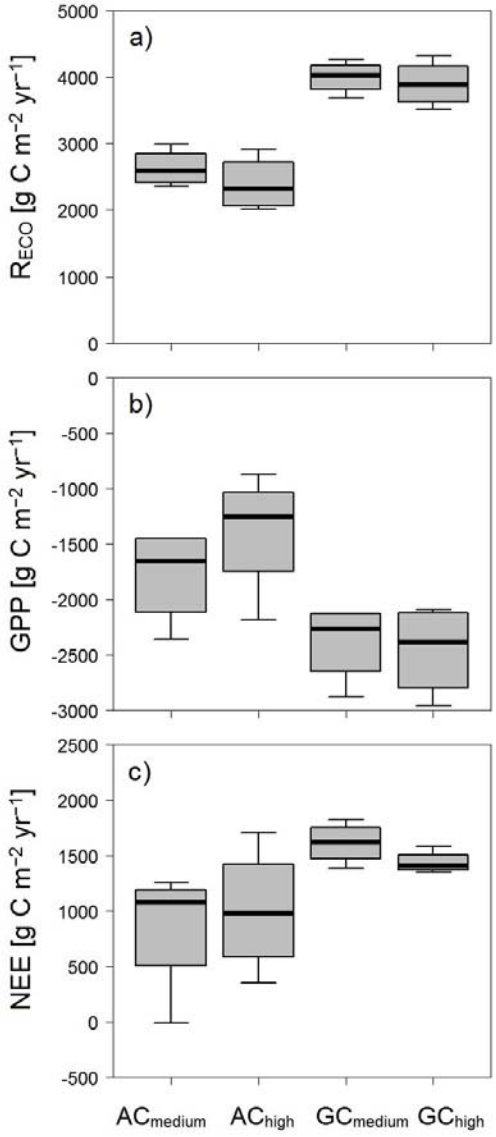
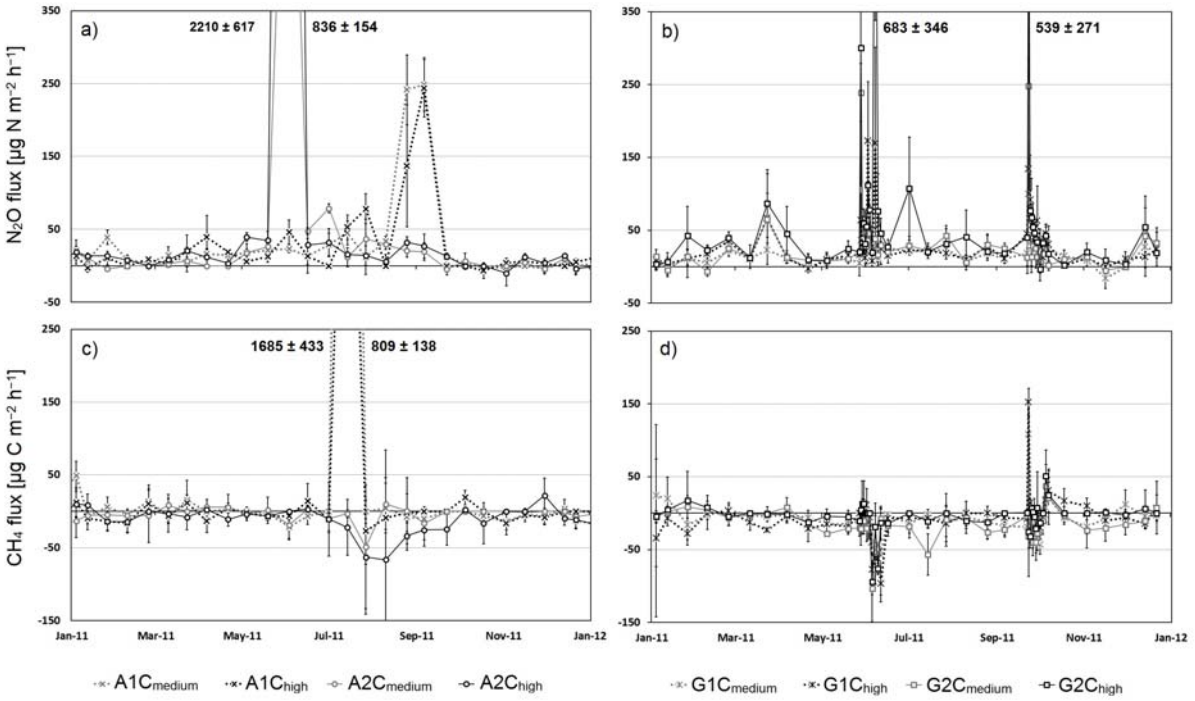


Fig. 6 Box plots of cumulative R_{ECO} (a), GPP (b) and NEE (c) for the two soil types and land-use types. Box plot showing median (central thick lines), 25% and 75% quartile ranges around the median (box width).



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1515 **Fig. 7** Time series of measured N_2O fluxes (a, arable land; b, grassland) and CH_4 fluxes (c, arable
1516 land; d, grassland) for the year 2011. Data from grassland plots (b,c) derived from Eickenscheidt et al.
1517 (2014b).

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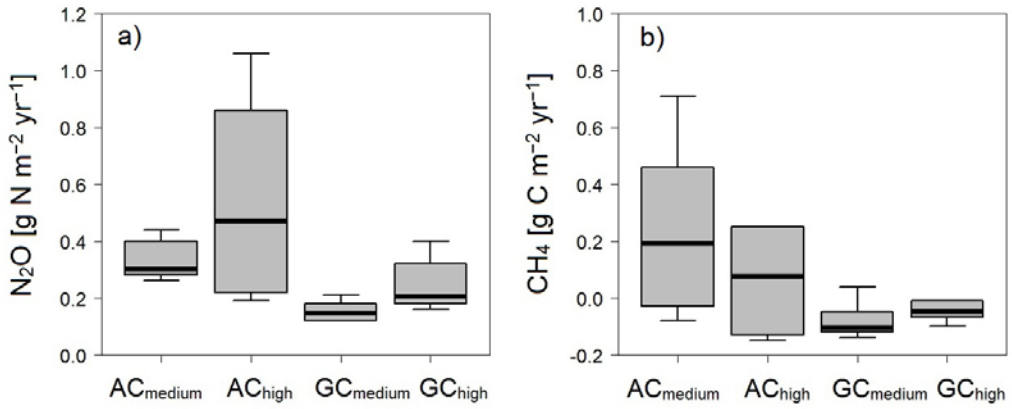

1519

1520 **Fig. 8** Box plots of cumulative annual N_2O emissions (a), and cumulative annual CH_4 emissions for the
1521 two soil types and land-use types. Box plot showing median (central thick lines), 25% and 75%
1522 quartile ranges around the median (box width).

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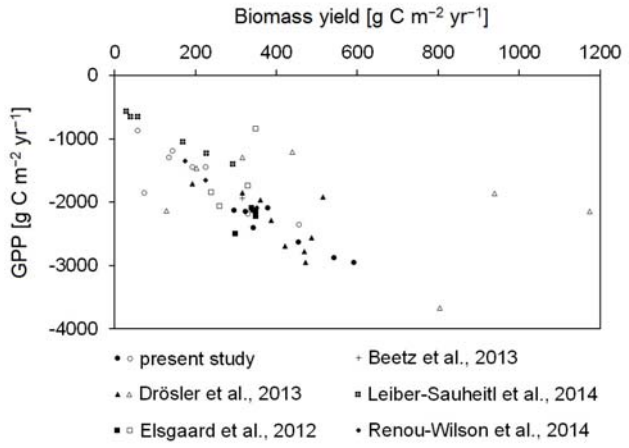
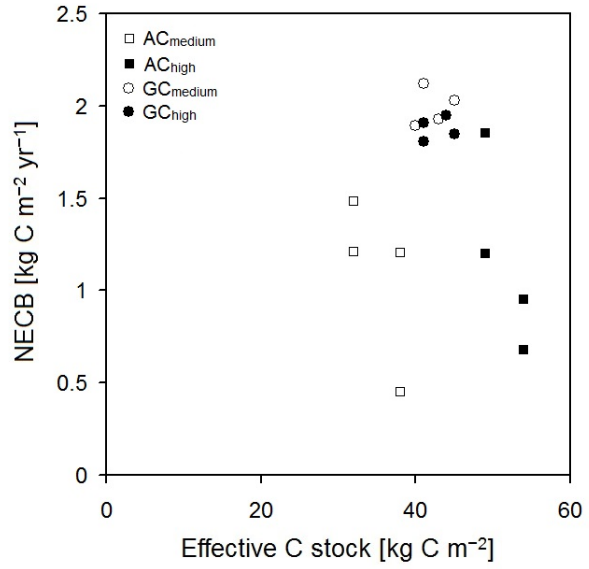


1524

1525 **Fig. 9** Relationship of GPP and biomass export from temperate peatlands. Filled symbols represents
1526 grassland sites (intensive and extensive), unfilled symbols represents arable lands.

1527



1528

1529 **Fig. 10** NECB plotted against the effective C stock, which is defined as the fraction of aerated carbon
1530 in the soil profile (according to Leiber-Sauheitl et al., 2014) (calculated NECB did not include CH_4
1531 losses).