



1 **Greenhouse gas emissions from fen soils used for forage**  
2 **production in northern Germany**

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10

11 **Abstract**

12 A large share of peatlands in northwest Germany is drained for agricultural purposes, thereby emitting  
13 high amounts of greenhouse gases (GHG). In order to quantify the climatic impact of fen soils in dairy  
14 farming systems of northern Germany, GHG exchange and forage yield were determined on four  
15 experimental sites which differed in terms of management and drainage intensity: a) rewetted and  
16 unutilized grassland (UG), b) intensive and ‘wet’ grassland (GW), c) intensive and ‘moist’ grassland  
17 (GM) and d) arable forage cropping (AR). Net ecosystem exchange (NEE) of CO<sub>2</sub> and fluxes of CH<sub>4</sub>  
18 and N<sub>2</sub>O were measured using closed manual chambers. CH<sub>4</sub> fluxes were significantly affected by  
19 groundwater level (GWL) and soil temperature, whereas N<sub>2</sub>O fluxes showed a significant relation to the  
20 amount of nitrate in top soil. Annual balances of all three gases, as well as the global warming potential  
21 (GWP), were significantly correlated to mean annual GWL. Two-year mean GWP, combined from  
22 CO<sub>2</sub>-C-equivalents of NEE, CH<sub>4</sub> and N<sub>2</sub>O emissions, as well as C input (slurry) and C output (harvest),  
23 was 3.8, 11.7, 17.7 and 17.3 Mg CO<sub>2</sub>-C-eq ha<sup>-1</sup> a<sup>-1</sup> for sites UG, GW, GM and AR, respectively  
24 (standard error (SE) 2.8, 1.2, 1.8, 2.6). Yield related emissions for the three agricultural sites were 201,  
25 248 and 269 kg CO<sub>2</sub>-C-eq (GJ net energy lactation (NEL))<sup>-1</sup> for sites GW, GM and AR, respectively  
26 (SE 17, 9, 19). The carbon footprint of agricultural commodities grown on fen soils depended on long-  
27 term drainage intensity rather than type of management, but management and climate strongly  
28 influenced interannual on-site variability. However, arable forage production revealed a high  
29 uncertainty of yield and therefore was an unsuitable land use option. Lowest yield related GHG  
30 emissions were achieved by a three-cut system of productive grassland swards in combination with a  
31 high GWL (long-term mean  $\leq$  20 cm below the surface).



1

2 **1 Introduction**

3 Natural peatland ecosystems act as long-term carbon (C) sinks as C in plant residues accumulates due to  
4 anoxic conditions and thus incomplete decomposition (Joosten & Clarke, 2002). Globally, the amount  
5 of C stored in peatlands is about 446 Pg (2 Pg in German peatlands) (Joosten, 2009), which is 24 %  
6 higher compared to the number of 359 Pg C stored in global forest vegetation, given by Dixon et al.  
7 (1994). The drainage of peatlands causes aerobic soil conditions, leading to accelerated mineralization  
8 of the soil organic matter (SOM) and an increased release of C and nitrogen (N) (Höper, 2002).  
9 Therefore, the natural sink for C and N is turned into a net source, converting drained peatlands to  
10 significant emitters of the greenhouse gases carbon dioxide (CO<sub>2</sub>) and nitrous oxide (N<sub>2</sub>O) (Kasimir-  
11 Klemetsson et al., 1997; Maljanen et al., 2003b, 2010). Simultaneously, the methane (CH<sub>4</sub>) emissions  
12 occurring under natural conditions are reduced to negligible levels (Roulet et al., 1993; van den Pol-van  
13 Dasselaaar et al., 1997; Maljanen et al., 2003a).

14 In Germany, peatlands cover around 1.67 million ha (Joosten, 2009), which corresponds to 4.7 % of the  
15 land area. Roughly 65 % of these peatlands are minerotrophic fens (Grosse-Brauckmann, 1997) and  
16 around 70 % is utilized for agricultural purposes (Röder & Osterburg, 2012). Peatland rich regions, as  
17 particularly northwest (NW) Germany (Lower Saxony, Schleswig-Holstein), show high shares of forage  
18 production and livestock units per ha of utilized agricultural area, which is attributed to a concentration  
19 of dairy farming (Röder & Osterburg, 2012). Consequently, there is a high demand for intensive forage  
20 production to ensure the supply of a high quality fodder. These management and cultivation practices  
21 require an intensive drainage and fertilization, leading to a continually increasing pressure on the  
22 utilization of German peatlands. The relevance of agricultural utilized peatlands for the national GHG  
23 budget is highlighted as only 5 % of the utilized agricultural area (Röder et al., 2011) but 50 % of the  
24 GHG emissions from agricultural soils (41.3 of 82.7 Tg CO<sub>2</sub>-equivalents (CO<sub>2</sub>-eq)) are attributed to  
25 peatlands drained for agriculture (UBA, 2014).

26

27 Restoration of cultivated organic soils has one of the greatest GHG mitigation potentials in agriculture  
28 (Smith et al., 2008). The reestablishment of the natural peatland functioning can only be achieved by  
29 abandoning the drainage based utilization, accompanied with a rewetting to natural hydrological  
30 conditions (Gorham & Rochefort, 2003; Höper et al., 2008; Zak et al., 2011). However, removing land  
31 from production provides maximum GHG mitigation, but might be rather an option for marginal lands  
32 than for regions with a high agricultural production value (Robertson et al., 2000). In those regions, it  
33 becomes fundamental to identify mitigation options that reduce GHG emissions without a distinct



1 reduction of the agricultural productivity (Smith et al., 2008). Furthermore, the objective of climate  
2 protection measures for these areas should focus on resource use efficiency, i.e. minimizing GHG  
3 emissions per unit of product instead of unit area (Oenema et al., 2014). Here, we will focus on the net  
4 exchange of the three biogenic trace gases CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O from fen soils in an intensive dairy  
5 farming region of northern Germany (Schleswig-Holstein) and relate their annual budgets to forage  
6 energy yield (net energy lactation, NEL) of the specific sites.

7 There are several publications about the climatic relevance of peatlands and their corresponding  
8 emission factors (Byrne et al., 2004; Alm et al., 2007a; Drösler et al., 2008; Oleszczuk et al., 2008;  
9 Couwenberg, 2009b; Maljanen et al., 2010). In recent years, advanced information about the GHG  
10 fluxes from German peatlands is emerging (Drösler, 2005; Couwenberg, 2011; Beetz et al., 2013; Beyer  
11 & Höper, 2014; Leiber-Sauheitl et al., 2014). Nevertheless, GHG data for agricultural managed fen soils  
12 in northern Germany is lacking and their function for forage production has not been considered in  
13 calculations about GHG mitigation. Therefore, the recommended strategy for GHG reductions from  
14 drained peatlands is the rewetting to natural conditions or extensification (Couwenberg et al., 2011;  
15 Beetz et al., 2013). However, in terms of reducing GHG emissions per unit forage produced, Renger et  
16 al. (2002) and Regina et al. (2014) report consistently that an average groundwater table of 30 cm below  
17 the soil surface enables high yielding grass cultivation and reduces the GHG emissions for a minimum  
18 of 40 %.

19 This study provides a full GHG balance as well as forage yields of fen soils in northern Germany in an  
20 intensive dairy farming region with different management strategies: a) rewetted and unutilized  
21 grassland (UG), b) intensive grassland ‘wet’ (GW), c) intensive grassland ‘moist’, (GM) and d) arable  
22 forage production (AR) and the assumptions that:

- 23 (i) rewetting leads to a decrease in CO<sub>2</sub> and N<sub>2</sub>O emissions but an increase in CH<sub>4</sub> emissions,
- 24 (ii) the GHG balances and C losses increase with land use intensity in the order UG > GW > GM >  
25 AR,
- 26 (iii) product related GHG emissions are higher for arable forage cropping on organic soils compared to  
27 grassland utilization,
- 28 (iv) wet but intensive grassland utilization (site GW) realizes lowest product related GHG emissions.

29



## 1 2 Material and Methods

### 2 2.1 Study area

3 The study was conducted in a huge lowland area of Schleswig-Holstein, the most northern state of  
4 Germany, at 54°21' N and 9°24' E. The long-term (1981 – 2010) mean annual temperature in this  
5 region is 8.7 °C and mean annual precipitation is 861 mm (Deutscher Wetterdienst (DWD), 2011). The  
6 region was shaped by meltwater at the end of the last ice age (Weichsel glacial stage) that flowed  
7 through the valleys originated by the previous ice age (Saale glacial stage). Thereby, river systems were  
8 formed and as a result of sea level and groundwater rise, deep fen soils developed that grew up to peat  
9 bogs at some locations (Blume & Brümmer, 1986). Since several centuries the area has been drained for  
10 agricultural utilization. Traditionally, the fen soils of the study area have been used as grasslands for  
11 forage production in dairy farms. In the past two decades about 15,000 ha of the region have been  
12 allocated for nature conservation purposes. In these areas the water levels were permanently raised and  
13 the agricultural utilization was extensified or abandoned (Rohman et al., 2008).

14 As a result of the ground level elevation as well as the status of the drainage system, the study area is  
15 irregularly drained, resulting in highly variable groundwater levels and thus intensity of peat  
16 degradation. According to these conditions, four sites were selected representing typical land use and  
17 drainage scenarios in this region. A rewetted and unutilized grassland site (UG) was chosen to evaluate  
18 the situation without agricultural activities. This site is located in a nature reserve area and was rewetted  
19 in 1991. There has been no utilization since 1998 and no fertilization since the rewetting. The  
20 vegetation of site UG is typical for wet and nutrient rich fallows, with a few dominant and productive  
21 species (Timmermann et al., 2006; Schrautzer et al., 2013). In contrast, the vegetation composition of  
22 the utilized grasslands (grassland 'wet', GW and grassland 'moist', GM) is dominated by species  
23 typical for intensively managed temperate grasslands (Table 1). The arable site (AR) was used as  
24 permanent grassland until conversion to silage maize production in 2007. In 2012, the cultivation  
25 changed to production of whole crop silage from spring barley and from spring wheat with undersown  
26 grass in 2013. The soil types of all sites are classified as *Histosols* according to FAO (2006).

27 The utilized grassland sites are fertilized with slurry from dairy cattle. Typically, this is conducted  
28 shortly before the beginning of the growing season in a range of 20 – 30 m<sup>3</sup> ha<sup>-1</sup> and subsequently after  
29 cutting events in a smaller range of 10 – 15 m<sup>3</sup> ha<sup>-1</sup> if another cutting is designated. The arable site  
30 received 35 and 18 m<sup>3</sup> ha<sup>-1</sup> of cattle slurry in 2011 and 2012, respectively. The slurry was deployed and  
31 incorporated into the top soil immediately before the sowing of the crops. In 2013, no slurry was  
32 applied. Additionally, the agricultural sites received mineral N fertilizers around the same dates as the



1 slurry application, which occurs mostly in the form of calcium ammonium nitrate (CAN), containing 27  
2 % of N. The total amounts of applied fertilizer N are displayed in Table 1.

3

## 4 **2.2 Site characteristics**

5 Air temperature, precipitation and photosynthetically active radiation (PAR) were measured at a climate  
6 station on site GW. When missing data occurred due to technical problems, data from a meteorological  
7 station of the DWD, located about 5 km from the sites, was used for gap filling. Soil temperatures in 5,  
8 10 and 15 cm depth of each site were continuously recorded every hour by soil temperature loggers  
9 (SL52T, IMEC, Heilbronn, Germany).

### 10 **2.2.1 Groundwater levels**

11 For continuous monitoring of groundwater levels (GWLs), four perforated PVC tubes ( $d = 3$  cm,  $l =$   
12 120 cm) were installed on each site in pairs at 5 and 15 m from the next drainage ditch. GWLs were  
13 recorded manually during every gas flux measurement campaign, leading to a minimum of one GWL  
14 record per week. For the calculation of mean annual GWLs, the recorded GWLs were linear  
15 interpolated to obtain daily values and to avoid overestimation of periods with more frequent  
16 measurements.

### 17 **2.2.2 Soil properties**

18 For monitoring of soil mineral N status, soil samples were taken fortnightly with a soil auger at a depth  
19 of 0 – 20 cm on each site. Nitrogen was extracted with 0.01 M  $\text{CaCl}_2$  (VDLUFA, 1997) and the  
20 concentrations of nitrate ( $\text{NO}_3^-$ ) and ammonium ( $\text{NH}_4^+$ ) of the extractions were analyzed  
21 photometrically with a dual channel continuous flow analyzer (San<sup>++</sup>, Skalar Analytical B.V., Breda,  
22 The Netherlands). Mineral N stocks per ha were calculated using the bulk density of the relevant sites.  
23 Bulk density was determined for the depths 5, 15, 25 and 45 cm according to DIN ISO 11272 (HBU,  
24 1998). The gravimetric water content of soil samples was estimated by oven drying at 105 °C. To  
25 calculate the contents and amounts of  $C_{\text{org}}$  and  $N_{\text{tot}}$  of each site, soil sampling was conducted twice a  
26 year at soil depths of 0 – 30, 30 – 60 and 60 – 90 cm. After oven drying (40 °C), samples were analyzed  
27 with an elemental analyzer (Vario Max CN, Elementar, Hanau, Germany). The soil pH was determined  
28 before and after the study period in 2011 and 2014 according to VDLUFA (1991).



## 1 **2.2.3 Herbage yield and forage quality**

2 To quantify the herbage yields, the above ground biomass (AGB) was cut shortly before harvest on  
3 three randomly selected spots with 0.25 m<sup>2</sup> at a height of 5 cm. The dry matter content of plants was  
4 determined after oven drying at 60 °C for 48 h. Subsequently, the material was grinded using a  
5 centrifugal mill equipped with a 1 mm sieve (Cyclotech mill, Tecator, Foss, Hillerød, Denmark). Forage  
6 quality parameters were estimated by near infrared reflectance spectroscopy (NIRS) (Baker & Barnes,  
7 1990). Therefore, each sample was scanned with a NIR-System 5000 monochromator (FOSS, Silver  
8 Spring, USA). The NIRS calibrations were based on a sample pool selected to represent the entire  
9 spectral and chemical variability for which N concentrations were directly measured with an elemental  
10 analyzer (Vario Max CN, Elementar, Hanau, Germany). Net energy lactation (NEL) as the feed energy  
11 content available for maintenance and milk production was estimated as a function of metabolizable  
12 energy (ME) and crude ash content (Weißbach et al., 1996), whereas ME was calculated from the  
13 contents of enzyme soluble organic matter, crude ash, crude fat and acid detergent fiber according to  
14 GfE (2008).

## 15 **2.3 Determination of GHG fluxes and balances**

### 16 **2.3.1 Flux measurements**

17 CH<sub>4</sub> and N<sub>2</sub>O fluxes were measured from April 2011 to March 2014 using closed manual chambers  
18 (Hutchinson & Mosier, 1981). Measurements were conducted weekly and in addition shortly after  
19 management practices like fertilization or tillage. At each site, eight PVC collars (d = 60 cm, h = 15 cm)  
20 were inserted 10 cm into the soil one week before the measurements started. To display gas fluxes for  
21 different GWLs at the same time, four collars were placed at 5 and 15 m from the next drainage ditch,  
22 respectively. When sites were harvested, the vegetation was removed from the collars. Site preparation  
23 measures were conducted in spring and the collars were shifted afterwards to obtain representative  
24 conditions. On site UG, a boardwalk was installed due to wet soil conditions and to avoid disturbances  
25 around the collars. For gas flux measurements, opaque PVC chambers (h = 35 cm, V = 0.1 m<sup>3</sup>) were  
26 used and chamber air samples were collected with a 30 ml syringe and stored in 12 ml pre-evacuated  
27 septum capped vials (Labco, High Wycombe, UK) (Glatzel & Well, 2008) 0, 15 and 30 min after  
28 chamber closure. Sampling was conducted between 09:00 and 12:00 to capture mean daily fluxes  
29 (Velthof & Oenema, 1995a; Petersen et al., 2012; van der Weerden et al., 2013). The samples were  
30 analyzed for concentrations of CH<sub>4</sub>, N<sub>2</sub>O and CO<sub>2</sub> with a gas chromatograph (7890a, Agilent  
31 Technology Inc., Santa Clara, CA, USA) equipped with a flame ionization detector (FID), electron  
32 capture detector (ECD) and thermal conductivity detector (TCD). Calibration of the gas chromatograph



1 was performed with a minimum of three certified gas standards. Samples were injected using an  
2 autosampler (222 XL, Gilson Inc., Middleton, WI, USA). Data processing was conducted with the  
3 software *Chem Station* (Version B.01.04, Agilent Technology Inc., Santa Clara, CA, USA).

4 The CO<sub>2</sub> exchange was determined according to the method of Drösler (2005). Elsgaard et al. (2012),  
5 Beetz et al. (2013) and Leiber-Sauheitl et al. (2014) present similar approaches. Here, static chambers  
6 with a diameter of 61 cm and a height of 35 were used. On each site three PVC collars were installed.  
7 Measurement campaigns were conducted during the period March 2012 until April 2014 in intervals of  
8 3 to 5 weeks. When harvest of the agricultural sites took place, the vegetation was removed from the  
9 collars and additional CO<sub>2</sub> measurements were carried out few days after harvest. In total, the CO<sub>2</sub>  
10 exchange was measured on 21, 28, 30 and 32 days at site UG, GW, GM and AR, respectively.  
11 Transparent and opaque chambers were used to measure the net ecosystem exchange (NEE) and the  
12 ecosystem respiration (R<sub>ECO</sub>), respectively. The chambers were connected to an infrared gas analyzer  
13 (LI-820, LI-COR Biosciences, Lincoln, NE, USA) and a data logger (CR 1000, Campbell Scientific,  
14 Logan, UT, USA). CO<sub>2</sub> concentration inside the chamber, temperature inside and outside the chamber  
15 and PAR outside the chamber were recorded every 5 s. Chambers were equipped with a fan to ensure  
16 homogenization of the atmosphere inside the chamber headspace. When the vegetation was higher than  
17 the chambers, extensions (h = 35 cm) were used. Measurement campaigns were conducted from sunrise  
18 until afternoon to comprise the whole daily range of PAR and soil temperature. Maximum enclosure  
19 times were 120 s for NEE and 300 s for R<sub>ECO</sub> measurements. Quality criteria for CO<sub>2</sub> measurements  
20 were changes of chamber temperature of more than 1.5 °C and a standard deviation of PAR more than  
21 10 % of average PAR. Measurements that exceeded these threshold values were discarded.

### 22 **2.3.2 Flux calculations**

23 Trace gas fluxes were calculated using linear regression for the change of gas concentration over time  
24 as it has been described in several other studies (e.g. Flessa et al., 1998; Chatskikh et al., 2008; Beetz et  
25 al., 2013). Since effects of temperature and pressure inside the chamber induce only minor uncertainties  
26 to the measured fluxes (Levy et al., 2011), these variables are often neglected in flux calculations  
27 (Chatskikh et al., 2008). However, to quantify the uncertainty in calculated CO<sub>2</sub> fluxes caused by a  
28 varying density of air as a function of temperature, CO<sub>2</sub> fluxes (n = 5546) were corrected for the mean  
29 temperature inside the chamber and compared to the uncorrected fluxes. On average, temperature  
30 correction reduced calculated fluxes by 6 % with a maximum reduction of 12 % at a very high  
31 temperature of 38 °C. As temperature was not measured inside the chambers for CH<sub>4</sub> and N<sub>2</sub>O flux  
32 measurements, the uncorrected CO<sub>2</sub> fluxes were used for further analyses to ensure methodological  
33 consistency.



1 For CH<sub>4</sub> and N<sub>2</sub>O, fluxes were accepted when the coefficient of determination (R<sup>2</sup>) of the linear  
 2 regression was ≥ 0.9 to ensure a high accuracy of measured fluxes. Measurements with R<sup>2</sup> < 0.9  
 3 occurred mainly when chamber concentrations were near ambient and the corresponding fluxes were  
 4 assumed to be 0. CO<sub>2</sub> concentrations of the gas samples were used as control to identify erroneous CH<sub>4</sub>  
 5 and N<sub>2</sub>O values. If the CO<sub>2</sub> concentration of a sample was not plausible (i.e. smaller than previous), the  
 6 fluxes of CH<sub>4</sub> and N<sub>2</sub>O were discarded from the dataset (Leiber-Sauheitl et al. 2014). For NEE and R<sub>ECO</sub>  
 7 measurements, all fluxes with plausible concentration changes over time were accepted, irrespective of  
 8 flux magnitude and the R<sup>2</sup> of linear regression (Alm et al., 2007b; Leiber-Sauheitl et al., 2014). To  
 9 avoid underestimation of CO<sub>2</sub> exchange by a diminishing concentration gradient between chamber  
 10 headspace and soil or plant, and thus decreasing fluxes (Davidson et al., 2002), only the part of linear  
 11 concentration change was used for flux calculation, which could be only 30 s for NEE measurements  
 12 with highly productive vegetation and high PAR.

### 13 2.3.3 CO<sub>2</sub> modelling

14 R<sub>ECO</sub> was estimated using a temperature-dependent flux model according to Lloyd & Taylor (1994):

$$15 \quad R_{ECO} = R_{ref} * \exp \left[ E_0 * \left( \frac{1}{T_{ref} - T_0} - \frac{1}{T - T_0} \right) \right] \quad (1)$$

16 where R<sub>ECO</sub> is the measured ecosystem respiration (g CO<sub>2</sub>-C m<sup>-2</sup> h<sup>-1</sup>), R<sub>ref</sub> is the respiration at reference  
 17 temperature (g CO<sub>2</sub>-C m<sup>-2</sup> h<sup>-1</sup>), E<sub>0</sub> is an activation-like parameter (K), T<sub>ref</sub> is the reference temperature  
 18 (283.15 K), T<sub>0</sub> is the temperature constant for the start of biological processes (227.13 K), and T is the  
 19 temperature with the best fit to the data of one measurement campaign. This could be either soil  
 20 temperature in 5 cm depth at the corresponding site or the air temperature from the weather station at  
 21 site GW. For modelling R<sub>ECO</sub>, R<sub>ref</sub> and E<sub>0</sub> were fitted plot based for each measurement campaign with  
 22 soil or air temperature, depending on the level of significance. If neither soil temperature nor air  
 23 temperature gave a significant relation to R<sub>ECO</sub> of a measurement campaign, the data was pooled with  
 24 that of one or two adjacent campaigns to obtain significant parameters for the R<sub>ECO</sub> model (Beetz et al.,  
 25 2013). However, for site UG it was in some cases not possible to calculate significant parameters.  
 26 Therefore, the dataset was separated into growing season and non-growing season according to Janssens  
 27 (2010) and all measurement campaigns of a season were pooled. By this approach, the temporal  
 28 resolution of the model was decreased, but the range of temperatures for which the model is valid, was  
 29 greatly increased. Nevertheless, for the agricultural sites it was necessary to consider the phenological  
 30 development of the plants and especially the effect of harvest in the model. When fitting the model per  
 31 campaign, the temperature range can be very narrow, which may lead to severe overestimations by the  
 32 R<sub>ECO</sub> model if the slope of regression is high and the temperature is above of the observed range.





1 Therefore, the highest measured value of the corresponding campaign was set as a threshold for  
 2 maximum  $R_{ECO}$ . Every modelled value exceeding that threshold was recessed. The fitted parameters  $R_{ref}$   
 3 and  $E_0$  were linear interpolated between the campaigns and  $R_{ECO}$  was modelled on an hourly basis using  
 4 the corresponding temperature. To calculate GPP, the modelled  $R_{ECO}$  at the time of NEE measurements  
 5 was subtracted from the measured NEE value.

6 GPP was modelled with PAR as input variable using the rectangular hyperbola of Michaelis & Menten  
 7 (1913):

$$8 \quad GPP = \frac{(GP_{max} * \alpha * PAR)}{(GP_{max} + \alpha * PAR)} \quad (2)$$

9 where GPP is the calculated gross primary production ( $g \text{ CO}_2\text{-C m}^{-2} \text{ h}^{-1}$ ),  $GP_{max}$  is the limit of carbon  
 10 fixation for infinite PAR ( $g \text{ CO}_2\text{-C m}^{-2} \text{ h}^{-1}$ ),  $\alpha$  is the initial slope of the regression curve or light use  
 11 efficiency ( $(g \text{ CO}_2\text{-C m}^{-2} \text{ h}^{-1}) (\mu\text{mol m}^{-2} \text{ s}^{-1})^{-1}$ ) and PAR is the average photon flux density of  
 12 photosynthetically active radiation ( $\mu\text{mol m}^{-2} \text{ s}^{-1}$ ) that was determined during the NEE measurement by  
 13 a quantum sensor (SKP 215, Skye Instruments, Llandrindod Wells, UK). PAR was corrected by a factor  
 14 of 0.92 as an absorption of 8 % by the transparent chambers was identified by own examinations.  $GP_{max}$   
 15 and  $\alpha$  were fitted plot based for each measurement campaign and linear interpolated between the  
 16 campaigns, assuming a consistent development of vegetation. However, as the plant biomass is  
 17 harvested,  $\text{CO}_2$  uptake is interrupted immediately. Therefore, the parameters of the preceding  
 18 measurement campaign, which was conducted only few days before harvest, were used until the cutting  
 19 and then set back to 0. The subsequent campaign was conducted within one week after the cutting to  
 20 capture the  $\text{CO}_2$  exchange of the recently harvested plants. GPP was modelled on an hourly basis using  
 21 measured PAR from the weather station at site GW.

### 22 **2.3.4 GHG and C balances**

23 As the net ecosystem exchange (NEE) of  $\text{CO}_2$  is the balance of  $\text{CO}_2$  uptake by plants (GPP) and the  
 24 autotrophic and heterotrophic respiration of plants and soil ( $R_{ECO}$ ) (Chapin et al., 2006), NEE was  
 25 calculated on an hourly basis as the sum of Eqs. (1) and (2):

$$26 \quad NEE = GPP + R_{ECO} \quad (3)$$

27 For further processing, GPP,  $R_{ECO}$  and NEE were calculated per hectare and summed up to daily values  
 28 ( $\text{kg CO}_2\text{-C ha}^{-1} \text{ d}^{-1}$ ). The site specific annual balances of the three components were calculated as the  
 29 average of the 365-days sums of the three replicates. Annual  $\text{CH}_4$  and  $\text{N}_2\text{O}$  balances were determined by  
 30 plot based linear interpolation between the measurement days and summation of daily values. Site  
 31 specific balances were calculated as average of the eight replicates. The global warming potential



1 (GWP) of a specific site indicates to which magnitude it contributes to global warming, based on the  
2 GHG balance for a certain period. GWP was calculated using the IPCC (2007) radiative forcing factors  
3 of the individual gases for a time horizon of 100 years. These are 25 for CH<sub>4</sub> and 298 for N<sub>2</sub>O related to  
4 CO<sub>2</sub> (CO<sub>2</sub>-equivalents (CO<sub>2</sub>-eq)). Additionally, anthropogenic C inputs and losses via slurry application  
5 and harvest were calculated as CO<sub>2</sub>-eq and included in the GWP (Beetz et al., 2013). Using the balances  
6 of CO<sub>2</sub>-C and CH<sub>4</sub>-C as well as the C import via slurry and C export via biomass harvest, the net  
7 ecosystem carbon balance (NECB) was calculated per site and year. For all C and GHG fluxes and  
8 balances, the atmospheric sign convention was applied, where all losses from the atmosphere into the  
9 ecosystem (site) are displayed as negative (the ecosystem acts as a sink) and all enrichments in the  
10 atmosphere are displayed as positive (the ecosystem acts as a source). This convention is transferred to  
11 the non-atmospheric fluxes like slurry application (negative) and biomass harvest (positive). GHG and  
12 carbon balances were calculated for the periods April 2012 – March 2013 and April 2013 – March  
13 2014.

## 14 **2.4 Statistical analyses**

15 The statistical software R (2014) was used to evaluate the data. Evaluation started with the definition of  
16 an appropriate statistical mixed model (Laird & Ware, 1982; Verbeke & Molenberghs, 2000). The data  
17 were assumed to be normally distributed and heteroscedastic due to the different sites and measurement  
18 periods. These assumptions are based on a graphical residual analysis. The statistical model included  
19 the site as a fixed factor. For daily CH<sub>4</sub> fluxes, GWL and soil temperature in 5 cm were modelled as  
20 covariates, whereas for N<sub>2</sub>O fluxes, the amount of nitrate in 0 – 20 cm soil depth was used. The year  
21 was regarded as a random factor. Also, the correlations of the measurement values due to the day of  
22 sampling were taken into account. Based on this model, an analysis of covariance (ANCOVA) was  
23 conducted to test for significant influences of the covariates.

24 For balances of CH<sub>4</sub>, N<sub>2</sub>O and CO<sub>2</sub>, as well as for the GWP, NECB and product related GHG  
25 emissions, a mixed model with the site as fixed factor and the year as random factor was defined in each  
26 case. Heteroscedasticity was modelled due to the different sites and measurement periods. An analysis  
27 of variance (ANOVA) was conducted to identify significant differences between the sites. For the yields  
28 of DM, C, N and NEL, the model was amplified by the year as a fixed factor instead of random factor.  
29 Furthermore, multiple contrast tests (Bretz et al., 2011) were conducted in order to identify significant  
30 differences between sites and years, respectively.

31 To evaluate the influence of GWL on the different trace gas balances and the total GWP, NECB and  
32 product related GHG emissions, mean annual GWL was added as a fixed factor to the model used for  
33 the *t*-test. This model was calculated with and without the interaction term of site and GWL, as well as



1 irrespective of the different sites. These three model types were compared referring to their Akaike  
2 information criterion (AIC) (Akaike, 1974) to assess which model gives the best estimate for the  
3 relation between GWL and the corresponding variable. For CH<sub>4</sub> and N<sub>2</sub>O balances, this procedure was  
4 conducted for mean annual GWL and, in terms of N<sub>2</sub>O, for mean annual soil nitrate.

5 For uncertainty analysis of the CO<sub>2</sub> model, a Monte Carlo simulation was conducted for each  
6 measurement plot and site. Therefore, model parameters with the same variation as the original values  
7 were randomly calculated for every measurement campaign or pooled dataset and new regressions with  
8 temperature (R<sub>ECO</sub>) and PAR (GPP) were fitted. Only regressions with realistic parameters were  
9 accepted (E<sub>0</sub> and R<sub>ref</sub> ≥ 0, α and GP<sub>max</sub> ≤ 0). This procedure was conducted 10,000 times, thus, 10,000  
10 different model outputs for R<sub>ECO</sub>, GPP and NEE were obtained. The variation of these randomly  
11 calculated model outputs represents the uncertainty that is caused by the chamber measurements and by  
12 the fitting and linear interpolation of different numbers of measurement campaigns per plot and year.  
13 Since this procedure is conducted for each plot, the uncertainty can be calculated as the sum of mean  
14 variance of the three plots per site and the variance resulting from averaging the three replicates. This  
15 uncertainty was used for comparison of means obtained by the original simulation. Leiber-Sauheitl et al.  
16 (2014) present a similar approach.

17

## 18 **3 Results**

### 19 **3.1 Weather conditions**

20 Comparing the air temperature of the study period to the long-term average (8.7 °C), the first period  
21 (2011/12) was warmer (9.6 °C), the second period (2012/13) was colder (8.1 °C) and the third period  
22 (2012/14) was warmer again (9.8 °C). The precipitation sums of the first two study periods (1012 mm  
23 in 2011/12 and 971 mm in 2012/13) were higher than the long-term annual precipitation sum (861 mm),  
24 whereas precipitation was lower in the third study period (821 mm).

25 Considerable differences between the three periods are consisting in days with mean temperatures  
26 below 0 °C (Fig. 1). While in the first and third winter only one period with 20 and 11 frost days,  
27 respectively, occurred, several freeze/thaw events and in total 58 days with mean temperatures below  
28 the freezing point appeared in the second winter. Therefore, in 2013 the vegetation period started about  
29 one month later than in 2012 and 2014. High precipitation events took place in August 2011, leading to  
30 a precipitation sum of 236 % the long-term average for this month (Fig. 1). Above-average precipitation  
31 also occurred in July 2012 (183 %), whereas in summer 2013 only 41 and 58 % of long-term average  
32 precipitation were registered in July and August, respectively.



## 1 **3.2 Groundwater levels**

2 Groundwater levels (GWLs) during the study period showed high variability between sites and years  
3 (Fig. 2). Highest fluctuations were recorded on sites GM and AR with the same minima and maxima of  
4 -88 and 2 cm, respectively. Variability was lower at sites UG and GW with minima of -56 (UG) and -65  
5 cm (GW) and maxima of 8 (UG) and 2 cm (GW) for the 3-year period. Also short-term fluctuations  
6 with GWLs close to the soil surface and deep water levels within a few days or weeks were more  
7 distinct at sites GM and AR. In summers 2011 and 2012, all sites showed high GWLs close to the  
8 surface and even periods of inundation at site UG, whereas in summer 2013 GWLs were considerably  
9 lower (Fig. 2).

10

## 11 **3.3 GHG fluxes**

### 12 **3.3.1 Methane (CH<sub>4</sub>)**

13 Daily methane fluxes were highest at site UG and low at the agricultural sites (Fig. 3). While the  
14 intensively drained sites GM and AR showed negligible CH<sub>4</sub> exchange, CH<sub>4</sub> fluxes were on a higher  
15 level at site GW with one distinct emission peak in April 2013. CH<sub>4</sub> emissions from site UG showed  
16 high spatial and temporal variability. Emissions increased for the first time in August 2011, followed by  
17 a continuous release of CH<sub>4</sub> until July 2013. Highest emission peaks were recorded in summer 2012 and  
18 after that high releases occurred in autumn 2012 and spring 2013. Remarkably, the CH<sub>4</sub> flux pattern at  
19 site UG changed substantially in July 2013 as emissions ceased and did not rise again until the end of  
20 the study period in spring 2014.

### 21 **3.3.2 Nitrous oxide (N<sub>2</sub>O)**

22 N<sub>2</sub>O fluxes during the 3-year period showed no distinct regularity at the unutilized site (UG), whereas  
23 the agricultural sites showed seasonal flux patterns with several emission peaks during spring, mainly  
24 occurring after N fertilization (Fig. 4). While emissions at site UG peak in May 2013, the highest N<sub>2</sub>O  
25 releases from site GW were observed in April 2012. Similar but more frequent emission peaks were  
26 recorded at site GM in April and May 2012 and 2013 and further distinct N<sub>2</sub>O releases from that site  
27 were observed in autumn and winter 2013. The most pronounced seasonality of N<sub>2</sub>O emissions was  
28 determined at the arable site (AR) with high releases at the beginning of each study period. Thereby, the  
29 emissions in May 2013 clearly exceeded those of the preceding two years.



### 1 3.3.3 Carbon dioxide (CO<sub>2</sub>)

2 The carbon dioxide exchange of the study sites was characterized by seasonal patterns of gross primary  
3 production (GPP) and ecosystem respiration (R<sub>ECO</sub>) with high exchange rates during the vegetation  
4 period and smaller fluxes between October and April (Fig. 5). Maximum CO<sub>2</sub> uptake rates were -176, -  
5 188, -228 and -320 kg CO<sub>2</sub>-C ha<sup>-1</sup> d<sup>-1</sup> for sites UG, GW, GM and AR, respectively (SE 9, 7, 17, 11).  
6 While this maximum C fixation took place in July 2013 at site UG, the two utilized grassland sites  
7 showed highest productivity in May 2012 before the first cutting. At site AR, maximum CO<sub>2</sub> uptake  
8 was modelled for the spring barley in June 2012. After the harvest of barley in August 2012, weeds  
9 remained that were eliminated by pesticides and mulched in September, so no CO<sub>2</sub> uptake could occur  
10 until emergence of newly seeded plants in May 2013. Maximum modelled CO<sub>2</sub> releases by R<sub>ECO</sub> from  
11 sites UG, GW, GM and AR were 156 (August 2012), 231 (May 2012), 216 (August 2012) and 259 kg  
12 CO<sub>2</sub>-C ha<sup>-1</sup> d<sup>-1</sup> (June 2012), respectively (SE 16, 6, 2, 11). Depending on the extent of daily GPP and  
13 R<sub>ECO</sub> fluxes, the sites can act as net source or sink for CO<sub>2</sub>. In total of two years (730 days), sites UG,  
14 GW, GM and AR acted as a CO<sub>2</sub> sink on 182, 156, 102 and 115 days, whereas they showed a net CO<sub>2</sub>  
15 release on 548, 574, 628 and 615 days, respectively (Fig. 5).

16

## 17 3.4 GHG balances, NECB and GWP

### 18 3.4.1 CH<sub>4</sub> and N<sub>2</sub>O balances

19 Over the three-year study period, mean annual CH<sub>4</sub> emissions were 55.1, 13.5, 0.9 and 1.8 kg CH<sub>4</sub>-C ha<sup>-1</sup>  
20 a<sup>-1</sup> for sites UG, GW, GM and AR, respectively (SE 17.2, 4.0, 0.5, 0.7). Highest annual CH<sub>4</sub> release  
21 occurred at site UG in the second year, while minimum budgets were determined for sites GM and AR  
22 in the third year (Table 2). However, due to the low fluxes at sites GM and AR, cumulated annual CH<sub>4</sub>  
23 emissions were not significantly different from zero ( $p > 0.05$ ). Sites GW and UG represented sources  
24 for CH<sub>4</sub> with significantly higher releases at site UG that also showed the highest variation in annual  
25 CH<sub>4</sub> budgets (Fig. 6a). Mean annual N<sub>2</sub>O balances of the four sites increased in the order UG, GW, GM  
26 and AR, accounting for 3.4, 6.5, 14.4 and 18.9 kg N<sub>2</sub>O-N ha<sup>-1</sup> a<sup>-1</sup>, respectively (SE 0.6, 0.9, 2.0, 1.1).  
27 Highest annual N<sub>2</sub>O emissions were recorded at site AR in the third year, whereas site UG released  
28 minimum amounts of N<sub>2</sub>O in the second year (Table 2). The high budgets of sites GM and AR showed  
29 high variation and thus, did not differ significantly ( $p = 0.18$ ) (Fig. 6b).



### 1 **3.4.2 CO<sub>2</sub> balances and NECB**

2 For the two years of CO<sub>2</sub> exchange measurement, mean annual NEE was 2.8, 8.0, 11.7 and 10.1 Mg  
3 CO<sub>2</sub>-C ha<sup>-1</sup> a<sup>-1</sup> for sites UG, GW, GM and AR, respectively (SE 2.5, 0.7, 1.2, 1.9) (Fig. 7a). Thus, all  
4 sites showed higher annual R<sub>ECO</sub> than GPP sums, with highest R<sub>ECO</sub> at site AR and lowest R<sub>ECO</sub> at site  
5 UG, both for the period 2013/14 (Table 2). Highest annual GPP was determined at site AR for 2013/14,  
6 whereas site GM showed lowest GPP during the same period. As for R<sub>ECO</sub> and GPP, both highest and  
7 lowest NEE occurred in 2013/14 at sites GM and UG, respectively (Table 2). As indicated by NECB,  
8 all sites were net C sources during the study period with mean annual losses of 2.8, 10.6, 15.7 and 15.0  
9 Mg C ha<sup>-1</sup> a<sup>-1</sup> at sites UG, GW, GM and AR, respectively (SE 2.6, 1.1, 1.4, 2.4) (Fig. 7b). Consistent  
10 with NEE, a higher range of NECB was assessed for the period 2013/14 with lowest C losses at site UG  
11 and highest losses at site AR (Table 2). The NEE and NECB of sites GW and AR did not differ  
12 significantly (Figs. 7a and b). However, mean NECB of site AR tended to be higher compared to site  
13 GW with  $p = 0.07$ .

### 14 **3.4.3 GWP**

15 The GWP combines the CO<sub>2</sub>-C-eqs of NEE, CH<sub>4</sub> and N<sub>2</sub>O emissions, as well as the anthropogenic C  
16 balances from slurry applications and biomass removals. For the study periods 2012/13 and 2013/14,  
17 mean annual GWP was 3.8, 11.7, 17.7 and 17.3 Mg CO<sub>2</sub>-C-eq ha<sup>-1</sup> a<sup>-1</sup> for sites UG, GW, GM and AR,  
18 respectively (SE 2.8, 1.2, 1.8, 2.6) (Fig. 7c). The lowest (site UG) as well as the highest GWPs (site  
19 AR) were observed for 2013/14 (Table 2). NEE dominated GWP at all sites with mean shares ranging  
20 from 59 % at site AR to 72 % at site UG. However, as no biomass removal occurred on site UG, this  
21 site also showed the highest shares of CH<sub>4</sub> and N<sub>2</sub>O, with each gas accounting for 14 % of the GWP on  
22 average of the two years. The GWPs of the agricultural sites were considerably influenced by the C  
23 balances of slurry inputs and harvested biomass, which accounted for 21, 23 and 27 % at sites GW, GM  
24 and AR, respectively.

## 25 **3.5 Crop yields and yield related GHG emissions**

### 26 **3.5.1 Biomass, carbon, nitrogen and energy yields**

27 For the grassland sites, all yield parameters were higher in 2012 than in 2013 (Table 3). While this  
28 reduction was significant for site GM, site GW showed no significant differences between years. At the  
29 arable site, significantly higher yields were obtained by spring wheat with undersown grass in 2013  
30 compared to spring barley in 2012. Site GM revealed significantly higher yields than site AR in 2012,  
31 while site GW did not differ to any other site in that year, except for N yield. In 2013, yields of sites



1 GM and AR showed no significant differences, while site GW had significantly lower yields than the  
2 other two sites, except for the N yield of site AR and the NEL yield of site GM. On average, site GM  
3 showed the highest yields, while lowest yields were observed on site GW, except for N yield, which  
4 was lowest on site AR. However, only N yield of sites GM and AR differed significantly.

### 5 **3.5.2 Yield related GHG emissions**

6 The annual GWP (Table 2) was related to the annual energy yields (Table 3) of the three agricultural  
7 study sites. While these yield related GHG emissions increased for site GM in the second year, they  
8 decreased for sites GW and AR (Table 4). On average of the two year study period, site GM did not  
9 differ significantly to the other sites, whereas site GW showed significantly lower yield related  
10 emissions than site AR.

11

## 12 **4 Discussion**

### 13 **4.1 CH<sub>4</sub> fluxes and balances**

14 Sites GM and AR showed negligible CH<sub>4</sub> fluxes and annual CH<sub>4</sub> budgets were not significantly  
15 different from zero. This is in accordance with other observations on intensively used peat soils that  
16 report low CH<sub>4</sub> emissions or even net uptake of CH<sub>4</sub> (Flessa et al., 1998; Maljanen et al., 2003a, 2004;  
17 Schäfer et al., 2012). The water table is the main controlling factor for CH<sub>4</sub> emissions from peat soils,  
18 particularly in absence of aerenchymus shunt species. A drainage depth of 20 – 30 cm is regarded as  
19 sufficient to inhibit the diffusion of high amounts of CH<sub>4</sub> into the atmosphere as CH<sub>4</sub> produced in the  
20 anoxic zone is oxidized by methanotrophs in the unsaturated zone (Couwenberg, 2009a; Schäfer et al.,  
21 2012). Accordingly, the low CH<sub>4</sub> fluxes at sites GM and AR can be explained by the high drainage  
22 intensity. However, a high GWL close to or above the soil surface did not enhance CH<sub>4</sub> production and  
23 emission at these sites (Fig. 8). A multiple linear regression model showed significant relations between  
24 log-transformed daily CH<sub>4</sub> fluxes and site ( $p < 0.001$ ), GWL ( $p < 0.001$ ) and soil temperature at 5 cm  
25 depth ( $p < 0.01$ ). Therefore, reactions on alterations of GWL and soil temperature differed between  
26 sites, probably as a consequence of long-term adaptation of methanogenic and methanotrophic  
27 communities to drainage intensity (van den Pol-van Dasselaar et al., 1997; Yrjälä et al., 2011). At site  
28 GW, CH<sub>4</sub> production potential was higher compared to sites GM and AR, leading to considerable CH<sub>4</sub>  
29 releases, especially when GWL and soil temperature were high, as for example in summer 2012 (Fig.  
30 3).



1 Conspicuous CH<sub>4</sub> peaks were detected at site UG in 2012 (Figs. 3 and 8) that were associated with high  
2 GWLs due to heavy rain fall in July and high soil temperatures due to a heat wave in late July and  
3 August (Figs. 1 and 2). These conditions likely favored a rapid expansion of the methanogenic  
4 community, more pronounced than in summer 2011 when GWLs were similarly high but temperatures  
5 were lower. Nykänen et al. (1998) reported that peat temperature controls CH<sub>4</sub> dynamics at high water  
6 tables, whereas the correlation is poor at low water tables. This is confirmed by the situation at site UG  
7 in summer 2013 when CH<sub>4</sub> emissions ceased as a consequence of low precipitation and water level  
8 drawdown in July and August, although soil temperatures were high. The subsequent GWL rise in  
9 autumn had no effect on CH<sub>4</sub> emissions, which remained low until the end of the study period. A  
10 possible explanation is that the methanogenic community was impaired by oxidative stress in summer  
11 (Görres et al., 2013) and did not recover due to low soil temperature when GWL rose (Bubier & Moore,  
12 1994). Knorr et al. (2008) reported that CH<sub>4</sub> production in a fen soil was retarded by experimental  
13 drought for up to several weeks after rewetting. Estop-Aragonés & Blodau (2012) observed a longer  
14 time lag until CH<sub>4</sub> production recovered after rewetting for more intense and longer dried fen peat but  
15 warmer conditions favored the recovery. Furthermore, the dry soil conditions in summer 2013 could  
16 have increased the methanotrophic community, leading to a CH<sub>4</sub> consumption potential in the  
17 subsequent months exceeding the production potential as methanotrophic bacteria react less sensitively  
18 to temperature changes than methanogenic bacteria (Dunfield et al., 1993). This is supported by the  
19 results of this study as the overall highest daily CH<sub>4</sub> uptakes were measured at site UG in summer and  
20 autumn 2013.

21 Annual CH<sub>4</sub> balances of the study sites are comparable to those recently reported for temperate  
22 European peat soils (Schäfer et al., 2012; Beetz et al., 2013; Leiber-Sauheitl et al., 2014). Annual  
23 balances were significantly related to site and mean annual GWL (both with  $p < 0.001$ ). Confirming the  
24 general understanding of CH<sub>4</sub> emission patterns (Couwenberg, 2009a), no significant CH<sub>4</sub> releases were  
25 observed for mean GWLs below -25 cm. At mean GWLs above -10 cm, CH<sub>4</sub> emissions were highly  
26 variable, with a minimum release of 28 and a maximum of 430 kg CH<sub>4</sub>-C ha<sup>-1</sup> a<sup>-1</sup> (Fig. 10a), which is  
27 typical for the high spatial variability of CH<sub>4</sub> fluxes (Waddington & Roulet, 1996; van den Pol-van  
28 Dasselaar et al., 1999). The low contribution of CH<sub>4</sub> emissions to the GWP of the three agricultural sites  
29 (Table 2) illustrates the minor importance of CH<sub>4</sub> in terms of GHG mitigation on utilized peat soils.  
30 However, Hahn-Schöfl et al. (2011) showed that degraded fen grasslands can emit huge amounts of  
31 CH<sub>4</sub> as a consequence of flooding when easily degradable fresh plant material is present. Therefore,  
32 inundation of sites with highly productive, energy rich grasses such as perennial ryegrass (*Lolium*  
33 *perenne*) bears the risk of enhanced CH<sub>4</sub> emissions, especially during summer. This should be  
34 particularly considered for site GW, where a significant CH<sub>4</sub> production potential could be observed.





1

## 2 4.2 N<sub>2</sub>O fluxes and balances

3 N<sub>2</sub>O emissions measured at the study sites were of similar magnitude as observed for other agricultural  
4 fen soils, for example in South Germany (Flessa et al., 1998), the Netherlands (van Beek et al., 2010;  
5 2011) or Denmark (Petersen et al., 2012) and conform to the range of N<sub>2</sub>O hotspots on European  
6 organic soils given by Leppelt et al. (2014). The N<sub>2</sub>O release from site UG represents the emissions  
7 without agricultural utilization in the study area. These were higher than reported for natural peatlands  
8 (Leppelt et al., 2014), which might be a result of GWL fluctuations (Figs. 2 and 10b), as background  
9 N<sub>2</sub>O emissions strongly depend on drainage intensity (van Beek et al., 2011). A multiple linear  
10 regression model for log-transformed daily N<sub>2</sub>O fluxes gave significant effects of site and the amount of  
11 nitrate in 0 – 20 cm soil depth (both with  $p < 0.001$ ) with highest fluxes measured at high soil nitrate.

12 Soil nitrate contents are enhanced by mineral fertilizer inputs on the one hand and mineralization and  
13 nitrification of organic N in soil organic matter (SOM) or organic fertilizers on the other hand. Several  
14 N<sub>2</sub>O emission peaks at the three agricultural study sites occurred subsequent to mineral fertilizer or  
15 slurry application, especially at site AR and in spring 2012 at all three sites (Fig. 4). High soil nitrate,  
16 exceeding the current N uptake capacity of vegetation can cause increased N<sub>2</sub>O production through  
17 denitrification, thus N fertilization often leads to enhanced N<sub>2</sub>O emissions for several days to weeks  
18 (Velthof & Oenema, 1995b; Bouwman et al., 2002; Grant et al., 2006). In addition, a nitrate surplus in  
19 soil promotes incomplete denitrification and increasing N<sub>2</sub>O/N<sub>2</sub> product ratios with the associated risk  
20 of N<sub>2</sub>O emissions (Firestone et al., 1980; Farquharson & Baldock, 2008; Senbayram et al., 2012). At  
21 site AR, strong N<sub>2</sub>O emission peaks occurred after fertilization in spring when vegetation was missing  
22 or seeded plants were emerging (Fig. 4).

23 Therefore, instead of relating annual N<sub>2</sub>O emissions to annual N balances, short-term N balances for  
24 about two week intervals were calculated for site AR and the vegetation periods 2012 and 2013 and  
25 related to the N<sub>2</sub>O balances of the same period. This was conducted by considering the N input by  
26 fertilizers as well as the N uptake by plants (Fig. 9). During the first weeks after fertilizer application, N  
27 surpluses of up to 99 kg ha<sup>-1</sup> occurred, leading to extremely high short-term N<sub>2</sub>O releases in some cases.  
28 The increasing N uptake in the subsequent periods was characterized by N balances ranging from -48 to  
29 12 kg N ha<sup>-1</sup> without significant N<sub>2</sub>O emissions. These findings confirm to a meta-analysis of van  
30 Groenigen et al. (2010), who found no differences in N<sub>2</sub>O emissions for negative or slightly positive N  
31 balances, but significantly increasing emissions for a surplus of 90 kg N ha<sup>-1</sup>. During the period  
32 2012/13, 73 % of N<sub>2</sub>O emissions at site AR occurred in April and May, while for the period 2013/14, 90  
33 % of the total annual N<sub>2</sub>O budget was emitted in May. Therefore, it can be concluded that in



1 combination with tilling, which might increase the availability of easily decomposable organic C for  
2 denitrifiers (Nykänen et al., 1995), fertilization of peat soils during periods with lacking N uptake  
3 capacity, bears the risk of substantial N<sub>2</sub>O emissions (Maljanen et al., 2003b; Regina et al., 2004).

4 After a second smaller fertilization peak at site AR in June 2013, N<sub>2</sub>O emissions were reduced to zero  
5 or even small uptakes of N<sub>2</sub>O were detected (Fig. 4), which can be explained by increased vegetation  
6 productivity. The growing plants act as competitor for nitrate to the denitrifiers, leading to complete  
7 denitrification as nitrate availability is strongly decreased. This was described for pristine (Roobroeck et  
8 al., 2009) or restored peatlands (Silvan et al., 2005) where N availability is usually limited (Martikainen  
9 et al., 1993). Our results suggest that on sites with very high N<sub>2</sub>O production potential, emissions can be  
10 eliminated by a continuous coverage of highly productive plants and prevention of fertilization when N  
11 uptake is limited. N<sub>2</sub>O uptake into soils is often linked to low mineral N and high moisture contents  
12 (Chapuis-Lardy et al., 2007). However, the small but continuous N<sub>2</sub>O uptakes at site AR, beginning in  
13 June 2013, were probably attributed to a high denitrification potential, stimulated by the excess of  
14 nitrate during May, and a shift to N<sub>2</sub>O consumption by denitrifiers when nitrate competition by plant  
15 roots increased (Roobroeck et al., 2009).

16 On average, N<sub>2</sub>O-N emissions from the agricultural study sites accounted for 2.2, 5.9 and 13.2 % of  
17 applied N for sites GW, GM and AR, respectively. The values for sites GW and GM fit well with those  
18 presented by van Beek et al. (2010) for grazed grasslands on organic soil in the Netherlands with  
19 comparable GWLs. Therefore, our results support the findings of van Beek et al. (2010), who argued  
20 that mean annual GWL should be used in addition to N input for estimating N<sub>2</sub>O emissions from  
21 organic soils, as the ratio of N<sub>2</sub>O emissions to N input increases with decreasing GWLs. However, our  
22 results illustrate that the type of management should be considered as well, as arable cropping can  
23 induce a disproportional increase of N<sub>2</sub>O emissions related to N input.

24 Drained organic soils are known to emit significant shares of their annual N<sub>2</sub>O budget during the winter  
25 period (Priemé & Christensen, 2001; Maljanen et al., 2003b), increasing with the number of freezing  
26 and thawing cycles (Regina et al., 2004). Thereby, N<sub>2</sub>O emissions are enhanced during freezing as well  
27 as thawing, since both processes release C into the soil, which is rapidly utilized by heterotrophic  
28 denitrifiers (Koponen et al., 2006). In contrast, emission peaks during winter were observed in the first  
29 and third year when only one period with negative temperatures occurred, but not in the second year,  
30 when more freezing and thawing cycles appeared (Figs. 1 and 4). The reason might be the deeper frozen  
31 soils in the first and third winters, as no snow cover was present, inducing higher C releases.

32 Annual N<sub>2</sub>O emissions were significantly related to mean annual GWL (Fig. 10b), which might be  
33 explained by increasing amounts of nitrate in top soil with increasing drainage intensity (Fig. 11a). As



1 the differences in soil nitrate could not be attributed to different N fertilization intensities (Table 1), the  
2 GWL seemed to control nitrification processes. Koops et al. (1997) emphasized that nitrification is an  
3 important process for N<sub>2</sub>O losses from peat soils, while Dowrick et al. (1999) stated that denitrification  
4 is the main source for N<sub>2</sub>O emissions from drained organic soils as the nitrate produced from peat  
5 mineralization is reduced in small-scale anaerobic porosity. However, both nitrification and  
6 denitrification processes likely contributed to N<sub>2</sub>O emissions as sites GM and AR show strong  
7 fluctuations in GWL (Fig. 2), which generally lead to a pronounced cycling of both processes and thus  
8 enhanced N<sub>2</sub>O release (Goldberg et al., 2010; Jørgensen & Elberling, 2012).

9

#### 10 **4.3 CO<sub>2</sub> exchange and NECB**

11 All four study sites were net C sources during the two years of CO<sub>2</sub> measurements (Table 2 and Fig. 7).  
12 Compared to IPCC (2014) emission factors for temperate organic soils, the sites showed NEE values  
13 above the given range for their respective land use categories. While the NEE of site AR was 9.0 Mg  
14 CO<sub>2</sub>-C ha<sup>-1</sup> a<sup>-1</sup> in 2012/13, which is within the 95 % confidence interval of 6.5 – 9.4 Mg CO<sub>2</sub>-C ha<sup>-1</sup> a<sup>-1</sup>  
15 given by IPCC (2014) for drained temperate croplands, it was above that range in 2013/14 (11.2 Mg  
16 CO<sub>2</sub>-C ha<sup>-1</sup> a<sup>-1</sup>). The NEE of sites GM and GW exceeded the intervals for nutrient-rich temperate  
17 grasslands that are deep-drained (5.0 – 7.3 Mg CO<sub>2</sub>-C ha<sup>-1</sup> a<sup>-1</sup>) or shallow-drained (1.8 – 5.4 Mg CO<sub>2</sub>-C  
18 ha<sup>-1</sup> a<sup>-1</sup>) in both years (Table 2). If the NECB is considered, the C losses of the agricultural sites were  
19 even higher, thus exceeding the upper values of IPCC emission factors for the respective land use  
20 categories by a factor of 2.0, 2.2 and 1.6 for sites GW, GM and AR, respectively. Moreover, the C loss  
21 from site UG clearly exceeded the average IPCC emission factor for rewetted and nutrient-rich  
22 temperate organic soils of 0.5 Mg CO<sub>2</sub>-C ha<sup>-1</sup> a<sup>-1</sup> in both years.

23 Recently published results for utilized organic soils in the same climatic region as the study area of this  
24 observation showed net C losses of 4.3 – 8.2 Mg CO<sub>2</sub>-C ha<sup>-1</sup> a<sup>-1</sup> for an intensively managed peat bog  
25 grassland in Germany (Beetz et al., 2013), 3.3 – 8.6 Mg CO<sub>2</sub>-C ha<sup>-1</sup> a<sup>-1</sup> for extensively managed  
26 grasslands on *histic Gleysol* in Germany (Leiber-Sauheitl et al., 2014) and 6.9 – 16.7 Mg CO<sub>2</sub>-C ha<sup>-1</sup> a<sup>-1</sup>  
27 for grassland and arable cropping on bog and fen soils in Denmark (Elsgaard et al., 2012). The highest  
28 value of 16.7 Mg CO<sub>2</sub>-C ha<sup>-1</sup> a<sup>-1</sup> represented a rotational grassland on fen soil, thus a comparable system  
29 to site AR in 2013/14, which showed a similar NECB of 17.7 Mg CO<sub>2</sub>-C ha<sup>-1</sup> a<sup>-1</sup>. However, the NEE of  
30 the Danish site was even higher (13.6 Mg CO<sub>2</sub>-C ha<sup>-1</sup> a<sup>-1</sup>) than at site AR (11.2 Mg CO<sub>2</sub>-C ha<sup>-1</sup> a<sup>-1</sup>),  
31 indicating that C removal by harvest from site AR was comparatively high. The permanent grassland  
32 sites studied by Elsgaard et al. (2012) showed C losses between 6.9 and 10.4 Mg CO<sub>2</sub>-C ha<sup>-1</sup> a<sup>-1</sup>. In  
33 conclusion, C losses of sites UG, GW and AR were at the upper end of literature values, while the



1 NECB of site GM clearly exceeded the given ranges. The comparatively high C losses of the study sites  
2 highlight the functioning of the study region as a considerable C source, underlining the need for  
3 mitigation strategies.

4 Seasonal variability of NEE on agricultural grasslands cannot only be explained by environmental  
5 parameters as their influence is often superposed by management activities like grassland cuttings  
6 (Wohlfahrt et al., 2008b). Land use intensity affects the NEE of ecosystems, as the frequency of  
7 biomass removals influences respiration processes as well as photosynthesis (Soussana et al., 2007).  
8 Generally, it is assumed that NEE increases with the number of cuttings, since GPP is reduced to almost  
9 zero for several days after harvest, while  $R_{ECO}$  can remain high, depending on the extent of soil  
10 respiration (Schmitt et al., 2010). At the studied grassland sites,  $R_{ECO}$  was often reduced by cutting  
11 events but not in the same degree as GPP, leading to sharp increases of NEE after harvest (Fig. 5). The  
12 effect of an increased number of grassland cuttings was especially pronounced at site GM, where four  
13 cuttings were conducted in the second year, compared to three cuttings in the first year. Thereby,  $R_{ECO}$   
14 was reduced to a greater extent than GPP, leading to a slightly increased NEE. However, at site GW the  
15 effect was different when the number of cuttings increased from two in the first to three in the second  
16 year. Here, a smaller  $R_{ECO}$  but slightly increased GPP resulted in a lower NEE in the second year. The  
17 same effect was visible for GPP when comparing sites GM and GW for a given year (Table 2). These  
18 results suggest that changing grassland management from two to three cuttings per year did not reduce  
19 total annual photosynthetic activity, while GPP could be diminished by four cuttings. However,  
20 irrespective of total number of grassland harvests, the first cuts were performed in May, the common  
21 time for intensively managed grasslands as the average growth rate is at its maximum (Parsons &  
22 Chapman, 2000). Before the first cut, the NEE of grasslands is mainly controlled by GPP (Wohlfahrt et  
23 al., 2008a). Shifting the first cut to June or July would therefore increase the total productivity of first  
24 growth period and extend the phase of net  $CO_2$  uptake. However, this is hardly compatible to intensive  
25 grassland management depending on profitability (McInerney, 2000) as forage quality would be too  
26 low. After a grassland cut it took several weeks until the sites showed net  $CO_2$  uptake again, often  
27 closely followed by the next cutting (Fig. 5). Therefore, the cutting regime strongly controlled the NEE  
28 of the agricultural grassland sites.

29 Unutilized peatland ecosystems can either be sources or sinks for  $CO_2$ , depending on variables like  
30 trophic status, peat temperature, water table (Bubier et al., 1998) or vegetation composition (Leppälä et  
31 al., 2011). As the difference between uptake (GPP) and release of  $CO_2$  ( $R_{ECO}$ ) is generally small,  
32 marginal changes of these parameters can invert the NEE of a peatland between different years (Bubier  
33 et al., 1999; Griffis et al., 2000; Arneth et al., 2002). At site UG, maximum daily GPP was observed in  
34 July, followed by a decrease in August, while  $R_{ECO}$  reached its maximum a few weeks later then



1 declined to a lesser extent. This was typical as the annual course of  $R_{ECO}$  is usually shifted by about one  
2 month compared to GPP (Lloyd & Taylor, 1994). Consequently, daily  $CO_2$  uptake reaches its maximum  
3 in spring or early summer and a net release of  $CO_2$  starts in late summer when vegetation becomes  
4 senescent and  $R_{ECO}$  exceeds GPP (Bellisario et al., 1998; Parsons & Chapman, 2000). A late cutting of  
5 vegetation could delay senescence and prolong the period of plant growth at site UG, which might  
6 reduce NEE. However, Beetz et al. (2013) observed that a single cutting event shifted a rewetted and  
7 extensively used peat bog grassland from a  $CO_2$  sink to a small source as annual GPP was reduced by  
8 more than annual  $R_{ECO}$ . This cutting was, however, conducted at the end of vegetation period and GPP  
9 did not rise again. The optimum time for a one-cutting grassland system in terms of maximizing GPP by  
10 avoiding early senescence might be in late July or early August to take advantage of both a highly  
11 productive primary growth and regrowth period. In addition, this was usually the period of lowest  
12 groundwater levels (Fig. 2), ensuring the viability of a grassland cutting as the limit for trafficability on  
13 fen soils is a GWL around -30 cm (Blankenburg et al., 2001). However, a potentially smaller NEE of a  
14 one-cut system might be offset by an increase in NECB due to biomass removal.

15 At site AR, the change of management with undersown grass in 2013 greatly influenced the courses and  
16 annual sums of GPP and  $R_{ECO}$  (Table 2 and Fig. 5). Both increased in the second year due to a  
17 continuous plant cover but with a larger increase of  $R_{ECO}$ , resulting in a higher NEE. As the C export by  
18 harvest also increased considerably (Table 3), the change of NECB was even greater than for NEE. In  
19 2012, no plants remained on the site after pesticide application and mulching in September, eliminating  
20 GPP and autotrophic respiration ( $R_a$ ). Due to a wet summer, harvest was conducted late and in spite of a  
21 high GWL, which induced soil compaction. In combination with the lack of water removal by plants,  
22 this led to inundation during autumn and winter. As a consequence, soil respiration was low during  
23 winter 2012/13 (Fig. 5). In contrast,  $R_{ECO}$  and GPP fluxes were higher in winter 2013/14 and  
24 considerably increased at the end of the study period due to highly productive new established grass, a  
25 lower GWL (Fig. 2) and higher temperatures (Fig. 1).

26 Several studies observed increasing  $CO_2$  emissions from peatland ecosystems with increasing drainage  
27 intensity (e.g. Moore & Knowles, 1989; Bubier et al., 1998; Drösler, 2005; Dinsmore et al., 2009).  
28 Since the variability of NEE for an individual agricultural site strongly depends on management  
29 (Wohlfahrt et al., 2008b) as described above, inter-site comparison is necessary to illustrate the effect of  
30 water level on NEE. On average of the four study sites and both years, NEE significantly increased by  
31 about  $220 \text{ kg } CO_2\text{-C } ha^{-1} a^{-1}$  per cm lowering of mean annual GWL (Fig. 12a). Moreover, our results  
32 suggest that arable cropping of peatlands did not lead to higher  $CO_2$  emissions per se, confirming recent  
33 observations from peatland sites in Germany (Drösler et al., 2013) and Denmark (Elsgaard et al., 2012).  
34 Despite a lower mean annual GWL on site AR (Table 1), NEE and NECB of sites AR and GM did not



1 differ significantly (Fig. 7). This can be explained by a lower  $R_{ECO}$  due to missing vegetation cover and  
2 water logging after harvest at site AR in the first year and a very high GPP due to undersown grass in  
3 the second year. Furthermore, Estop-Aragonés et al. (2012) argue that in compacted peat soils with high  
4 bulk densities and ash contents, oxygen penetration is reduced compared to less compacted soils,  
5 resulting in lower air filled porosity and soil respiration. Due to the higher peat degradation of site AR  
6 (Table 1), this could partly explain the similar NEE of sites AR and GM.

7 While Aurela et al. (2007) reported that a drought period in a Finnish sedge fen increased  $R_{ECO}$  and thus  
8 NEE, Leppälä et al. (2011) concluded that the difference in NEE between wet and dry years for natural  
9 peatlands in Finland resulted from alterations of GPP rather than  $R_{ECO}$ . For the dryer second year of our  
10 observations,  $R_{ECO}$  of site UG was lower than in the first year, while GPP decreased only marginally  
11 (Table 2). However, comparing only July and August, the period with greatest difference in GWLs  
12 between the years (-9.2 cm in 2012 and -36.6 cm in 2013; Fig. 2),  $R_{ECO}$  was almost the same (6.9 and  
13 6.8 Mg CO<sub>2</sub>-C ha<sup>-1</sup> in 2012 and 2013, respectively), which is in line with results presented by  
14 Parmentier et al. (2009) and Muhr et al. (2011). GPP slightly increased in the drier year (-7.2 and -7.6  
15 Mg CO<sub>2</sub>-C ha<sup>-1</sup> in 2012 and 2013, respectively). As main reason for differences in NEE between the  
16 two years, the weather conditions in spring could be identified. In 2012, the growing season, calculated  
17 by the method of Janssens (2010), started on 20 March, while it was delayed by more than one month in  
18 2013 to 23 April. As a result of different weather conditions, cumulated  $R_{ECO}$  for April and May was 4.1  
19 Mg in 2012 and only 2.2 Mg CO<sub>2</sub>-C ha<sup>-1</sup> in 2013. Besides, GPP was -3.2 Mg for April and May 2012  
20 and -2.3 Mg CO<sub>2</sub>-C ha<sup>-1</sup> in 2013. These differences cannot be explained by mean GWL for the two  
21 months (-4.9 cm in 2012 and -8.4 cm in 2013) as the different weather conditions were the dominating  
22 parameter. Thus, respiration processes were stimulated more than plant productivity by the earlier start  
23 of growing season, indicating that shorter winter periods potentially increase the risk of higher C losses  
24 from peatland ecosystems. Griffis et al. (2000) studied the NEE of a subarctic fen and concluded that  
25 the phenological stage of vegetation relative to the climatic conditions is important for interannual  
26 variability of NEE. In conclusion, the mean GWL of single years cannot be solely used to predict the  
27 variability of NEE at the same site or between sites with different management as climatic and  
28 management effects can be of dominating importance.

29

#### 30 **4.4 Global warming potential**

31 The global warming potential (GWP) of the four study sites increased in the same order as NEE and  
32 NECB. However, the difference between sites GW and AR was significant for GWP whereas it was not  
33 significant for NEE and NECB (Fig. 7). This can be explained by significantly higher N<sub>2</sub>O emissions at



1 site AR (Fig. 6b). NEE mainly controlled the GHG balances, accounting for 72, 69, 66 and 59 % of the  
2 GWP on sites UG, GW, GM and AR, respectively. In addition, the balances of C export via harvest and  
3 C import via slurry contributed considerably to the GWP of the agricultural sites, accounting for 21, 23  
4 and 27 % for sites GW, GM and AR, respectively, indicating a higher share of anthropogenic C fluxes  
5 with higher land use intensity. Compared to other observations or reviews of peatland GHG emissions  
6 in northern or temperate Europe, the GWP of the study sites was at the upper end of presented emission  
7 factors (Nykänen et al., 1995; Langeveld et al., 1997; Kasimir-Klemmedtsson et al., 1997; Alm et al.,  
8 2007a; Oleszczuk et al., 2008; Maljanen et al., 2010; Drösler et al., 2013).

9 Site UG showed a significantly lower GWP compared to the agricultural sites, supporting the  
10 assumption that rewetting of drained organic soils reduces their climatic footprint (Höper et al., 2008;  
11 Beetz et al., 2013). The lower GWP of site UG was a result of missing C losses through harvest and  
12 reduced CO<sub>2</sub> and N<sub>2</sub>O emissions that could mainly be attributed to the high GWLs (Fig. 12b),  
13 outweighing the higher CH<sub>4</sub> release (Fig. 6a). A linear regression for all four sites and both years gave a  
14 significant increase of GWP for about 410 kg CO<sub>2</sub>-C-eq ha<sup>-1</sup> a<sup>-1</sup> per cm lowering of mean annual GWL  
15 (Fig. 12b). The higher slope compared to NEE (Fig. 12a) was a result of N<sub>2</sub>O emissions, significantly  
16 increasing with drainage intensity as well (Fig. 10b). However, as CH<sub>4</sub> emissions tended to increase  
17 exponentially when water levels were close to the soil surface (Fig. 10a), the slope might decline or  
18 even invert for a mean annual GWL around or above 0 (Augustin & Joosten, 2007). Therefore, the  
19 intercept of ~2 Mg CO<sub>2</sub>-C-eq ha<sup>-1</sup> a<sup>-1</sup> should not be over-interpreted. A mean annual GWL of about 10  
20 cm below the soil surface is often referred to as an optimum scenario for mitigating GHG emissions  
21 from peatlands, as CO<sub>2</sub> emissions are greatly reduced or even negative (i.e. CO<sub>2</sub> uptake) and CH<sub>4</sub> fluxes  
22 are hampered by the small oxic horizon (e.g. Couwenberg et al., 2011). However, this is not only  
23 controlled by mean annual GWL, but equally by groundwater fluctuations (Dinsmore et al., 2009).  
24 Thus, the relatively high GWP of site UG (3.8 Mg CO<sub>2</sub>-C-eq ha<sup>-1</sup> a<sup>-1</sup>) in spite of a high mean annual  
25 GWL (Table 1) suggests that a further increase and stabilization of water levels might be necessary to  
26 reduce the climatic impact of that site. The lack of natural, peat forming mire vegetation (Table 1)  
27 supports this assumption as the GWP of natural or rewetted reed and sedge fens is assumed to be around  
28 1 Mg CO<sub>2</sub>-C-eq ha<sup>-1</sup> a<sup>-1</sup> (Couwenberg et al., 2011; Drösler et al., 2013).

29

#### 30 4.5 Yield related emissions

31 To assess the climatic footprint of the agricultural study sites, their function in terms of forage and milk  
32 production has to be considered in addition to area related GHG emissions. On average of two years,  
33 site GW represented the most climate-efficient forage production system of the three sites, whereas site



1 AR caused the greatest GHG emissions relative to energy yield (Table 4). Observations of greenhouse  
2 gas emissions from arable forage cropping systems at two sites on mineral soil in northern Germany  
3 resulted in yield related emissions between -18 and 32.5 kg CO<sub>2</sub>-C-eq (GJ NEL)<sup>-1</sup>, including all  
4 emissions during crop production, transport and storage (Herrmann et al., 2014). Hence, the field based  
5 emissions at the study sites presented here, demonstrate that forage produced on fen soils is burdened  
6 with many times higher GHG emissions compared to forage from mineral soils of the same region.

7 The high yield related emissions of site AR were mainly attributed to the low energy yield of barley in  
8 the first year, resulting from wet conditions in summer and thus a delayed harvest with low quality for  
9 milk production. In addition, the site was only partially harvested due to high soil moisture, thus, the  
10 ‘true yield’ per ha was even lower than given in Table 3. Moreover, the maize in 2011 could not be  
11 harvested at all due to above-average precipitation in August and September (Fig. 1). Therefore, arable  
12 forage production on fen soils of the study area is associated with a high uncertainty of yield in wet  
13 years, which, considering the high GHG emissions, makes it an inappropriate type of management from  
14 both an economic and environmental point of view. Underlining this conclusion, the management of  
15 site AR was changed in 2013 with undersown grass, increasing the certainty of yield as the time of  
16 harvest became more flexible. However, despite a high yield in 2013, yield related emissions remained  
17 higher compared to site GW as a result of a very high GWP (Table 2).

18 Comparing the two grassland sites, the four-cut system of site GM in 2013 showed the highest and the  
19 three-cut system of site GW in 2013 induced lowest yield related GHG emissions. In addition, the two-  
20 cut system of site GW in 2012 had higher yield related emissions than the three-cut systems of both  
21 sites. Therefore, a three-cut grassland in combination with a preferably high GWL represented the most  
22 climate-efficient management system at the studied fen soils. On average of both years, the energy yield  
23 of site GW was 19 % lower compared to site GM, while the GWP was 34 % lower. This difference was  
24 only significant for GWP. Thus, the effect of a raised water level can be assumed to be greater for GHG  
25 emission reduction than for yield reduction. This is in line with results of Renger et al. (2002), who  
26 reported that for a mean GWL of -30 cm, 90 % of optimum plant output can be reached, while GHG  
27 emissions can be reduced for 40 – 50 % of maximum emissions. These values were obtained by a water  
28 regime model and represent an optimum scenario, indicating that further potential exists to improve the  
29 climate efficiency of forage production on site GW. Reasons for reduced productivity on poorly drained  
30 soils could include the loss of sown species in favor of undesirable species with increasing sward age  
31 (Hopkins & Green, 1979) and a lower soil warming in spring due to high soil moisture, resulting in  
32 delayed plant growth (Tyson et al., 1992). The first aspect was evident in increasing shares of creeping  
33 bentgrass (*Agrostis stolonifera*) and water foxtail (*Alopecurus geniculatus*) at site GW, indicating the  
34 need for occasional resowing of productive species like perennial ryegrass (*Lolium perenne*).





1 None of the conventional management options can be regarded as sustainable in terms of peat  
2 conservation as each type of utilization associated with peatland drainage led to peat mineralization  
3 (Joosten & Clarke, 2002; Renger et al., 2002). The ongoing subsidence due to peat loss might change  
4 the utilization structure in future as sites become wetter and some areas might need to be extensified or  
5 abandoned, opening potentials for GHG mitigation. This was recently evident at site GW, where only  
6 two cuts could be realized in the wet years 2011 and 2012.

7

## 8 **5 Conclusions**

9 Long-term drainage intensity was the most important controlling factor for GHG emissions from the  
10 studied fen soils. NEE dominated the GHG balances of all sites and as assumed, considerable  
11 differences in GHG fluxes and balances were observed among the sites. After 20 years of rewetting (site  
12 UG), emissions of CO<sub>2</sub> and N<sub>2</sub>O were significantly lower while significantly higher amounts of CH<sub>4</sub>  
13 were emitted compared to the agricultural sites. Also the GWP of site UG was significantly reduced.  
14 However, the site still acted as a C source and showed substantial N<sub>2</sub>O emissions, indicating that  
15 rewetting had not yet restored the natural peatland functioning as a sink for C and a negligible source  
16 for N<sub>2</sub>O. Restoration progress could be promoted by a year-round stabilization of GWL close to the soil  
17 surface. In the current state, a mulching of vegetation in summer might increase total annual  
18 productivity by avoiding early senescence and thus reduce C losses.

19 Arable forage production (site AR) did not induce higher C losses compared to intensive grassland  
20 management and only showed a significantly higher GWP than the wet grassland site (GW) as the  
21 influence of drainage intensity was of dominating importance. However, interannual on-site variability  
22 was additionally affected by management and climatic factors. The beginning of growing season was  
23 identified as a critical period, with higher CO<sub>2</sub> losses occurring with an early start of vegetation period.  
24 Yield related GHG emissions increased with increasing drainage and land use intensity in the order  
25 GW, GM and AR, with a significant difference between sites GW and AR.

26 As arable cropping was associated with a high uncertainty of yield and a high GWP, this type of  
27 management was identified as unsuitable for forage production on fen soils. The wet grassland site  
28 (GW) realized lowest yield related emissions due to a significantly lower GWP in combination with a  
29 non-significantly reduced energy yield compared to sites GM and AR. Thus, this study demonstrated  
30 that there is huge potential for GHG mitigation in intensively utilized peatland areas of northern  
31 Germany which could be realized without eliminating traditional forage production. Reducing the land  
32 use intensity (low N fertilization, late first cut) of increasingly inundating areas as a consequence of peat



1 loss, could further enhance GHG mitigation and additionally promote nature conservation purposes  
2 (particularly meadow bird protection).

3

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14

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1 Table 1. Soil and land use characterization of the experimental sites (UG: unutilized grassland, GW: grassland ‘wet’, GM:  
 2 grassland ‘moist’, AR: arable land). Numbers in brackets represent standard deviation.

Site	UG (1 ha)	GW (3 ha)	GM (3.5 ha)	AR (2.2 ha)
Peat depth (cm)	180	420	360	280
C <sub>org</sub> (%) <sup>a</sup>	35.0 (2.6)	37.4 (3.9)	17.9 (2.9)	13.3 (1.9)
C/N <sup>a</sup>	17.7 (1.0)	15.7 (0.6)	12.4 (0.4)	12.2 (0.2)
Ash (%) <sup>a</sup>	36.8 (11.7)	33.6 (6.3)	68.7 (2.3)	74.0 (4.2)
Bulk density (g cm <sup>-3</sup> ) <sup>a</sup>	0.20 (0.05)	0.32 (0.07)	0.54 (0.08)	0.67 (0.09)
C stock (Mg ha <sup>-1</sup> ) <sup>a</sup>	215 (57)	361 (82)	289 (45)	266 (38)
N <sub>min</sub> (kg ha <sup>-1</sup> ) <sup>b</sup>	20.8 (8.8)	44.7 (22.7)	73.1 (37.3)	65.3 (31.4)
NO <sub>3</sub> -N/NH <sub>4</sub> -N <sup>b</sup>	0.10 (0.16)	0.25 (0.27)	0.67 (0.61)	2.55 (3.34)
Soil moisture (kg kg <sup>-1</sup> ) <sup>b</sup>	2.84 (0.44)	2.36 (0.61)	1.15 (0.30)	0.79 (0.16)
pH (CaCl <sub>2</sub> ) <sup>c</sup>	4.58 (0.13)	4.41 (0.18)	5.06 (0.13)	5.31 (0.29)
Groundwater level (cm) <sup>d</sup>	-10.9 (3.5)	-21.4 (4.6)	-33.0 (9.4)	-39.4 (4.2)
Fertilization (kg N ha <sup>-1</sup> a <sup>-1</sup> ) <sup>e</sup>	—	300 (240 – 400)	260 (230 – 320)	150 (130 – 170)
Type of fertilizer <sup>f</sup>	—	cattle slurry, CAN, ASN	cattle slurry, CAN	cattle slurry, DAP, CAN
Dominant plant species	Purple small-reed ( <i>Calamagrostis canescens</i> ), Reed canary grass ( <i>Phalaris arundinacea</i> ), Common rush ( <i>Juncus effusus</i> )	Perennial ryegrass ( <i>Lolium perenne</i> ), Rough bluegrass ( <i>Poa trivialis</i> ), Creeping bentgrass ( <i>Agrostis stolonifera</i> )	Italian ryegrass ( <i>Lolium multiflorum</i> ), Perennial ryegrass ( <i>Lolium perenne</i> ), Rough bluegrass ( <i>Poa trivialis</i> )	Maize ( <i>Zea mays</i> ), Barley ( <i>Hordeum vulgare</i> ), Wheat ( <i>Triticum aestivum</i> ), Perennial ryegrass ( <i>Lolium perenne</i> )

3 <sup>a</sup> Given values are for 0 – 30 cm soil depth. C<sub>org</sub> and C/N: mean value from biannual samplings during the period May 2011  
 4 – March 2014 (n = 7). Bulk density and C stock: mean value of soil samples taken in May 2013 (n = 4). Ash content: mean  
 5 value from samples taken in October 2013 (n = 4). <sup>b</sup> Mean value of mineral nitrogen (NO<sub>3</sub><sup>-</sup> and NH<sub>4</sub><sup>+</sup>) and gravimetric soil  
 6 moisture content in 0 – 20 cm soil depth from biweekly samplings during the period April 2011 – April 2014 (n = 73). <sup>c</sup>  
 7 Mean value of two samplings in the beginning (May 2011) and in the end (July 2014) of the study (n = 8). <sup>d</sup> Mean value of  
 8 linear interpolated weekly measurements in the period April 2011 – March 2014 (n = 4). <sup>e</sup> Sum of applied nitrogen from  
 9 organic and mineral fertilizers on average of 2011, 2012 and 2013 and the range between the years. <sup>f</sup> CAN = calcium  
 10 ammonium nitrate, ASN = ammonium sulphate nitrate, DAP = diammonium phosphate.

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1 Table 2. Annual budgets of CO<sub>2</sub> exchange (R<sub>Eco</sub>, GPP and NEE), CH<sub>4</sub> and N<sub>2</sub>O fluxes, net ecosystem carbon balance  
 2 (NECB) and global warming potential (GWP) for 100 years (IPCC, 2007) for different study periods (each period is April –  
 3 March). NECB is calculated from NEE and CH<sub>4</sub>-C as well as slurry-C and harvested C. The GWP includes CO<sub>2</sub>-C-  
 4 equivalents of NEE, CH<sub>4</sub>-C, N<sub>2</sub>O-N, slurry-C and harvested C. Small deviations in NEE are caused by rounding. Values are  
 5 annual sums and standard errors (in brackets).

Site	Period	R <sub>Eco</sub> (Mg CO <sub>2</sub> -C ha <sup>-1</sup> )	GPP (Mg CO <sub>2</sub> -C ha <sup>-1</sup> )	NEE	CH <sub>4</sub> (kg C ha <sup>-1</sup> )	N <sub>2</sub> O (kg N ha <sup>-1</sup> )	NECB (Mg C ha <sup>-1</sup> )	GWP
					(Mg CO <sub>2</sub> -C-eq ha <sup>-1</sup> )		(Mg CO <sub>2</sub> -C-eq ha <sup>-1</sup> )	
UG	2011/12	—	—	—	38.6 (10.6)	3.3 (1.2)	—	—
	2012/13	20.7 (2.3)	-16.8 (3.1)	3.9 (3.8)	99.5 (47.8)	2.3 (0.4)	4.0 (3.9)	5.1 (4.3)
	2013/14	17.9 (0.8)	-16.3 (0.9)	1.7 (1.2)	27.0 (9.7)	4.5 (1.1)	1.7 (1.2)	2.5 (1.3)
GW	2011/12	—	—	—	6.6 (1.7)	4.7 (1.1)	—	—
	2012/13	26.6 (0.2)	-17.5 (0.3)	9.1 (0.4)	16.5 (2.8)	8.9 (2.0)	13.0 (1.0)	14.3 (1.2)
	2013/14	24.7 (0.8)	-17.8 (0.5)	6.9 (0.9)	17.3 (11.7)	5.9 (1.0)	8.3 (1.1)	9.2 (1.3)
GM	2011/12	—	—	—	1.1 (0.6)	11.4 (3.0)	—	—
	2012/13	29.6 (1.1)	-18.3 (0.4)	11.4 (1.1)	1.5 (0.8)	12.5 (1.9)	15.7 (1.3)	17.3 (1.5)
	2013/14	24.9 (1.2)	-13.0 (0.4)	12.0 (1.3)	0.2 (0.9)	19.3 (4.7)	15.7 (1.5)	18.2 (2.9)
AR	2011/12	—	—	—	0.8 (1.8)	20.0 (1.8)	—	—
	2012/13	24.7 (1.3)	-15.7 (0.9)	9.0 (1.5)	4.5 (0.7)	15.3 (1.4)	12.2 (1.8)	14.2 (1.9)
	2013/14	33.3 (2.2)	-22.1 (0.7)	11.2 (2.3)	0.2 (0.3)	21.6 (2.1)	17.7 (3.0)	20.5 (3.3)

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1 Table 3. : Annual yields of dry matter (DM), carbon (C), nitrogen (N) and net energy lactation (NEL) for the three  
 2 agricultural utilized study sites and two years. Different capital letters indicate significant differences between the sites for a  
 3 particular year. Different lowercase letters indicate significant differences between the years for a particular site ( $p < 0.05$ ).  
 4 Values in brackets are standard errors ( $n = 3$ ). Crops at site AR were summer barley (2012) and summer wheat with  
 5 undersown grass (2013).

Site	Year	DM		C		N		NEL	
		(Mg ha <sup>-1</sup> a <sup>-1</sup> )		(Mg ha <sup>-1</sup> a <sup>-1</sup> )		(kg ha <sup>-1</sup> a <sup>-1</sup> )		(GJ ha <sup>-1</sup> a <sup>-1</sup> )	
GW	2012	10.7 (1.2)	ABa	4.9 (0.6)	ABa	234 (31)	Ba	63.9 (7.0)	ABa
	2013	8.2 (0.4)	Aa	3.7 (0.2)	Aa	218 (8)	Aa	53.6 (3.2)	Aa
	Mean	9.5 (0.8)	A	4.3 (0.4)	A	226 (15)	AB	58.7 (3.9)	A
GM	2012	13.1 (0.3)	Bb	5.9 (0.2)	Bb	335 (7)	Cb	78.8 (1.7)	Bb
	2013	10.0 (0.4)	Ba	4.5 (0.2)	Ba	274 (17)	Ba	66.1 (3.0)	ABa
	Mean	11.5 (0.6)	A	5.2 (0.3)	A	305 (13)	B	72.5 (2.7)	A
AR	2012	8.2 (0.5)	Aa	3.7 (0.2)	Aa	107 (11)	Aa	47.5 (2.6)	Aa
	2013	14.6 (1.6)	Bb	6.5 (0.7)	Bb	296 (34)	ABb	88.1 (8.2)	Bb
	Mean	11.4 (1.3)	A	5.1 (0.6)	A	202 (38)	A	67.8 (8.2)	A

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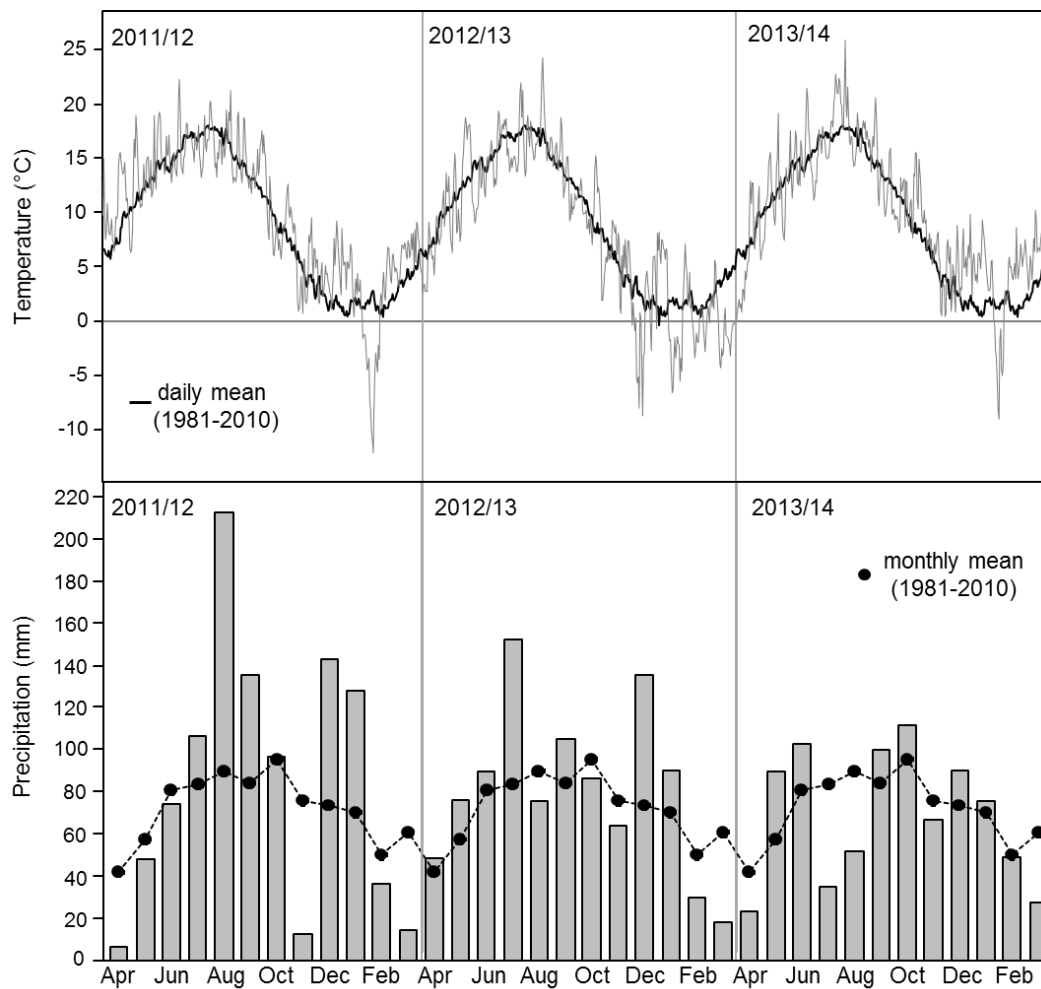


- 1 Table 4. Annual GHG balances (CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O fluxes, slurry-C and harvested C) of the three agricultural study sites in  
 2 relation to energy yield (net energy lactation, NEL). Different capital letters indicate significant differences between the sites  
 3 (p < 0.05). Values in brackets are standard errors (n = 3).

Period	Site					
	GW		GM		AR	
	kg CO <sub>2</sub> -C-eq (GJ NEL) <sup>-1</sup>					
2012/13	231 (25)		220 (5)		301 (18)	
2013/14	172 (10)		276 (12)		236 (21)	
Mean	201 (17)	A	248 (9)	AB	269 (19)	B

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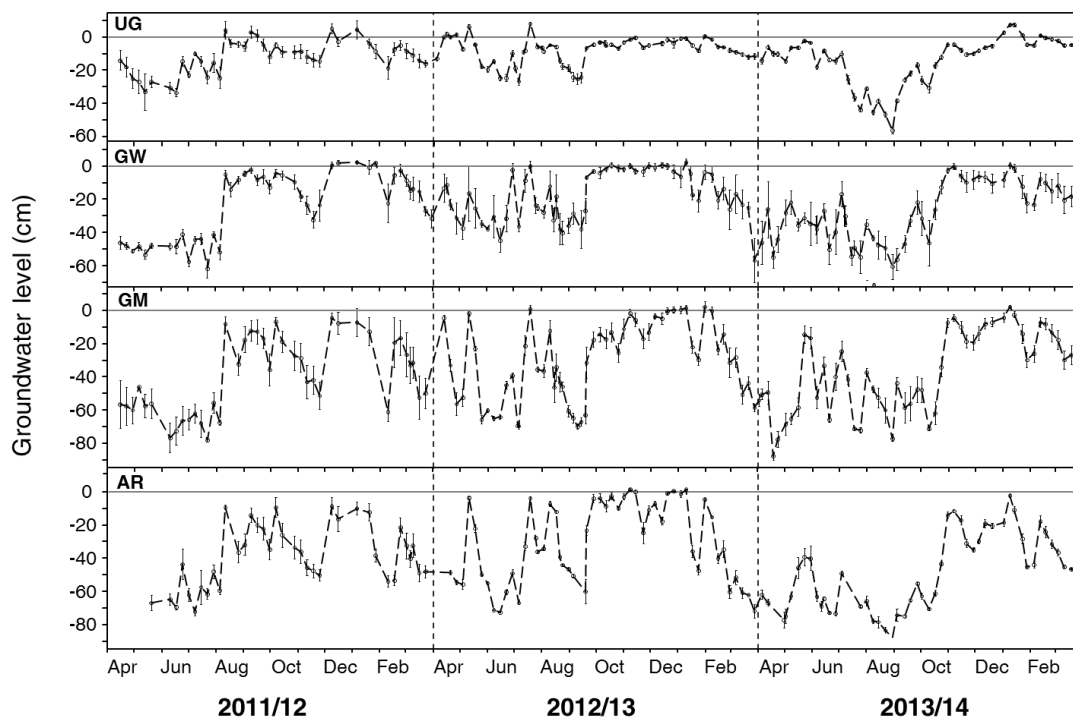


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3 Figure 1. Daily mean air temperatures (grey line) and monthly mean precipitation sums (grey bars)  
4 during the study period (April 2011 – March 2014) compared to the long-term averages.

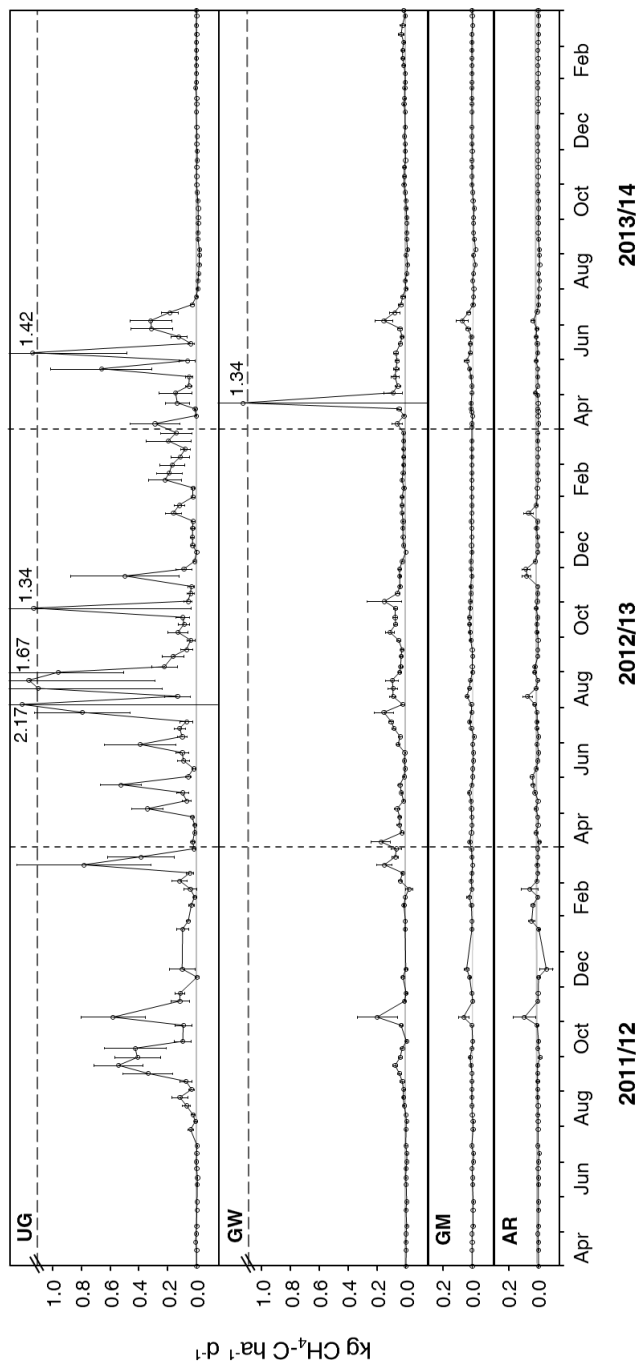
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3 Figure 2. Development of groundwater levels (GWLs) at the four study sites during the study period  
4 (April 2011 – March 2014). Displayed are mean values  $\pm$  standard errors of the manually recorded  
5 GWLs during gas flux measurements ( $n = 4$ ).

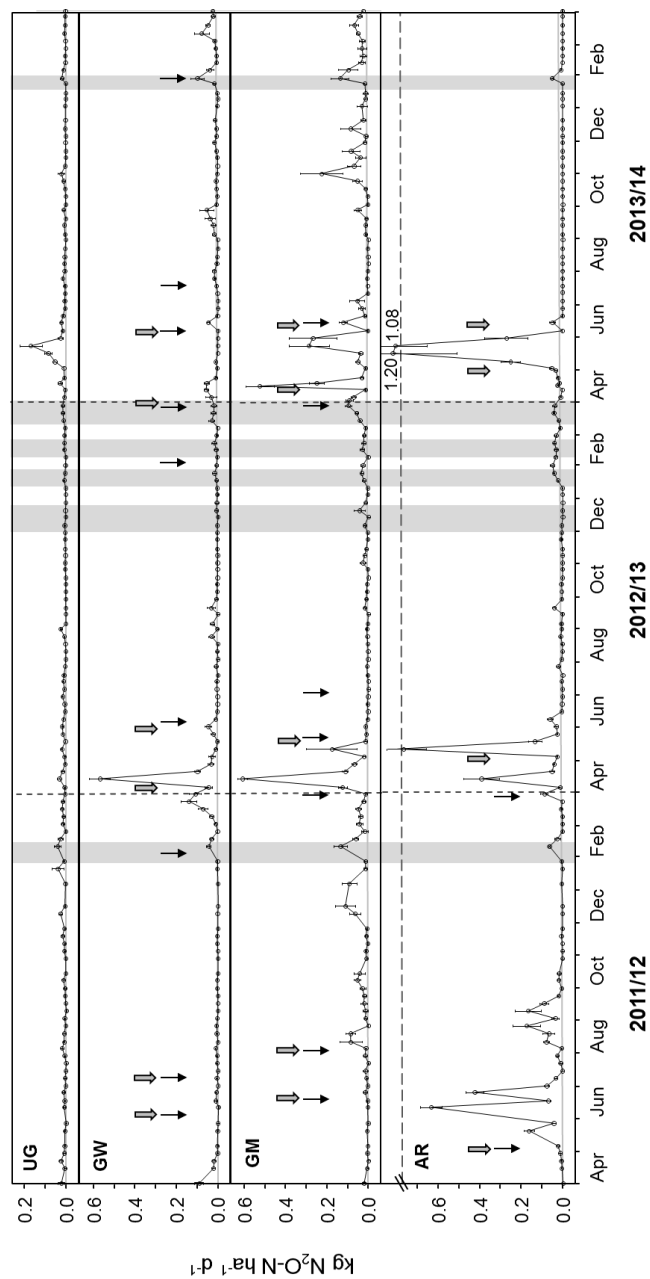


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3 Figure 3. CH<sub>4</sub> exchange at the four study sites during the study period (April 2011 – March 2014).

4 Values are displayed as mean ± standard error (n = 8). Note the broken y-axis for sites UG and GW.



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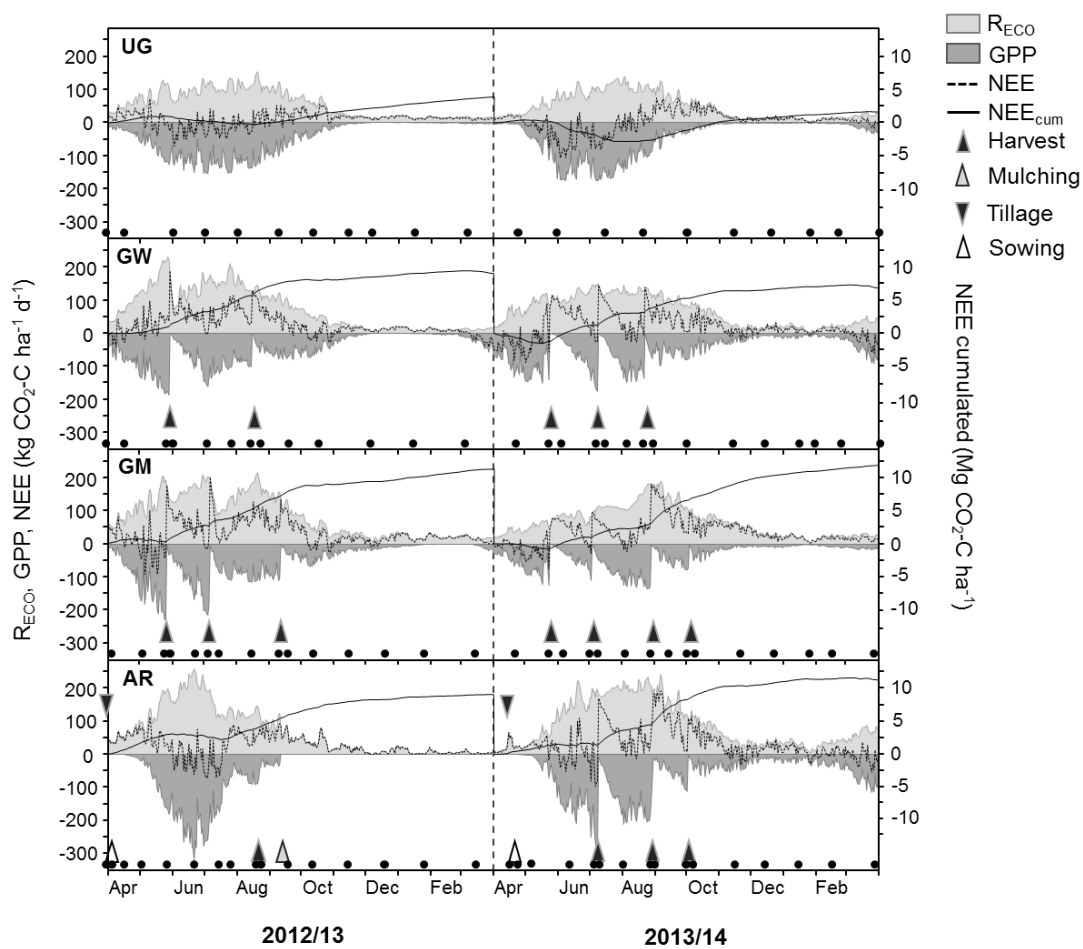
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3 Figure 4. N<sub>2</sub>O exchange at the four study sites during the study period (April 2011 – March 2014).

4 Values are displayed as mean ± standard error (n = 8). Note the broken y-axis for site AR. Arrows

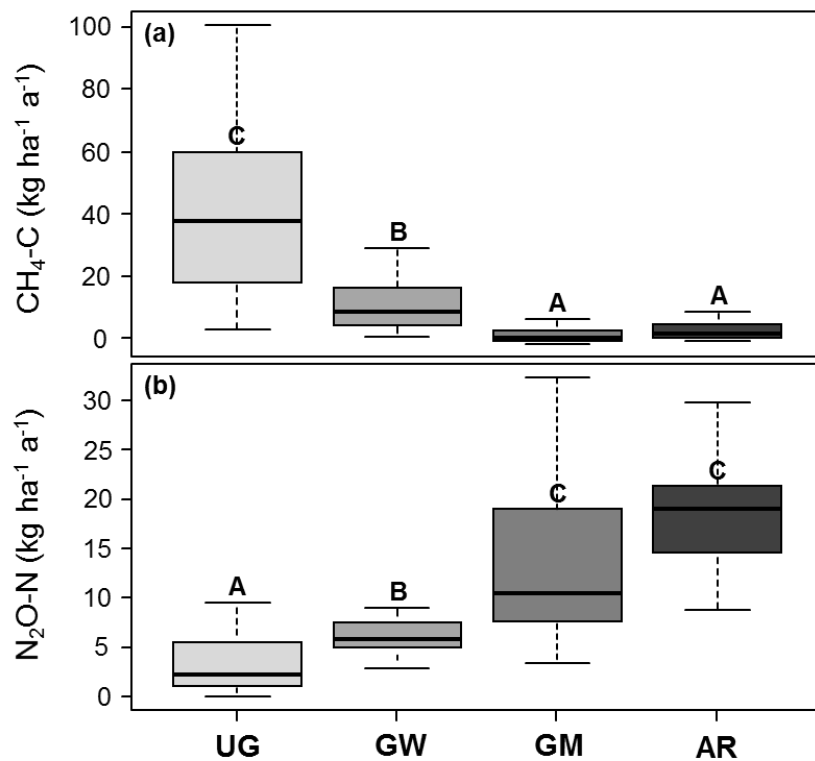
5 indicate applications of slurry (black) and mineral nitrogen fertilizer (grey). Grey background represents

6 periods with mean daily temperatures below 0 °C.



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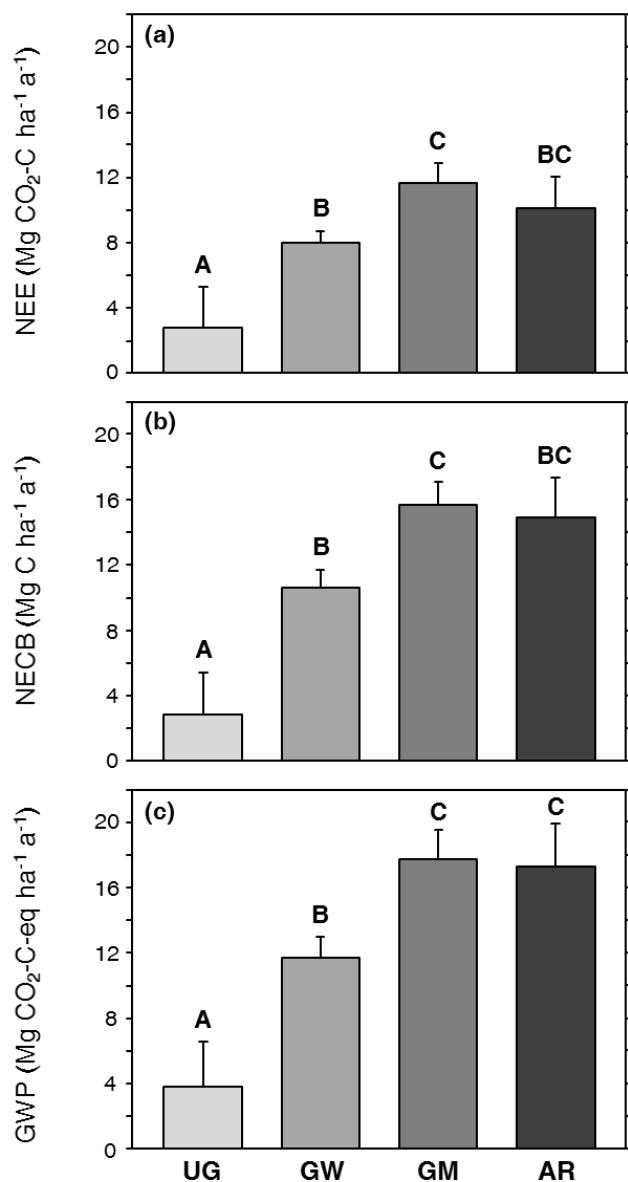
3 Figure 5. CO<sub>2</sub> exchange at the four study sites during a two-year study period (April 2012 – March  
 4 2014). Values are displayed as daily means of the model output (n = 3). The black continuous line  
 5 shows the cumulated NEE for one year. The black dots represent CO<sub>2</sub> measurement campaigns.



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3 Figure 6. Annual emissions of CH<sub>4</sub> (a) and N<sub>2</sub>O (b) at the four study sites combined for three years  
4 (April 2011 – March 2014) of measurement (n = 24). Different capital letters indicate significant  
5 differences between the sites (p < 0.05).

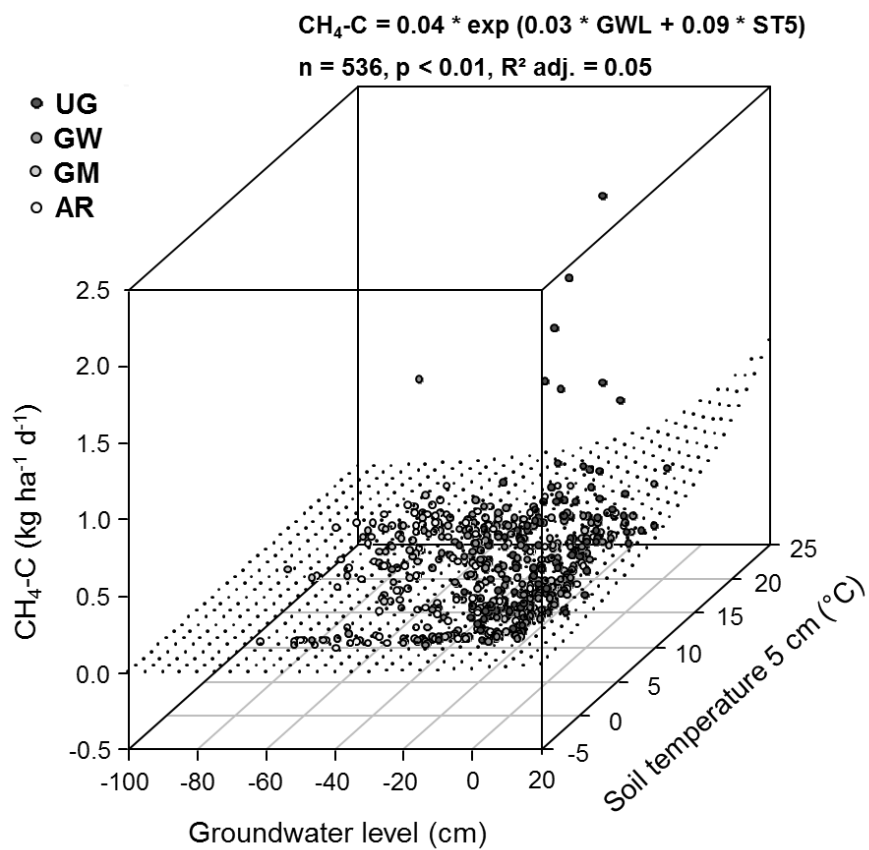


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3 Figure 7. Mean annual budgets of net ecosystem exchange (NEE) of CO<sub>2</sub> (a), net ecosystem carbon  
4 balance (NECB) (b), and global warming potential (GWP) (c) at the four study sites for the period April  
5 2012 – March 2014 (n = 6). Error bars represent standard error. Different capital letters indicate  
6 significant differences between the sites (p < 0.05).





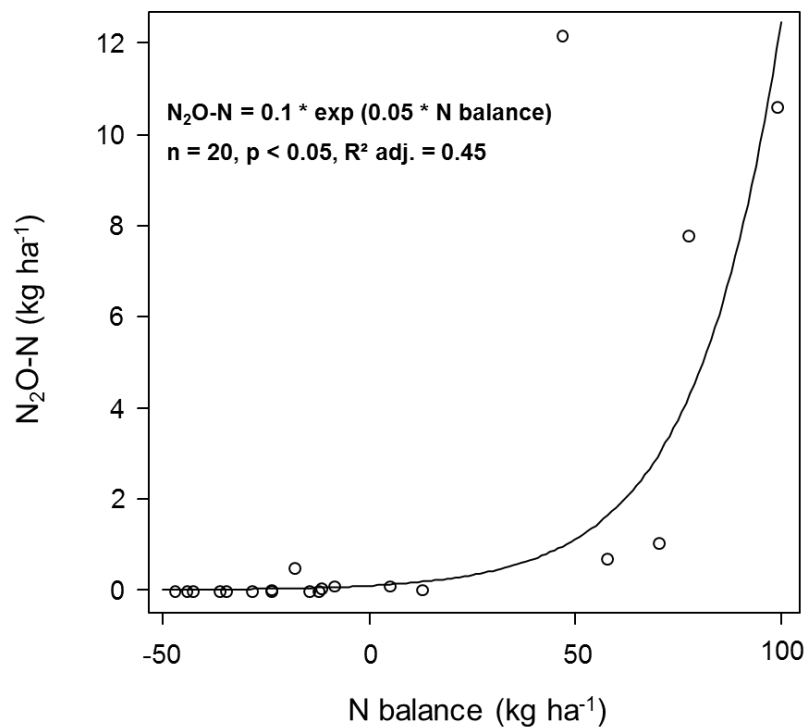
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3 Figure 8. Relationship of daily CH<sub>4</sub> fluxes to groundwater level and mean daily soil temperature at 5 cm

4 depth. GWL in the equation is groundwater level (cm) and ST5 is soil temperature at 5 cm depth (°C).

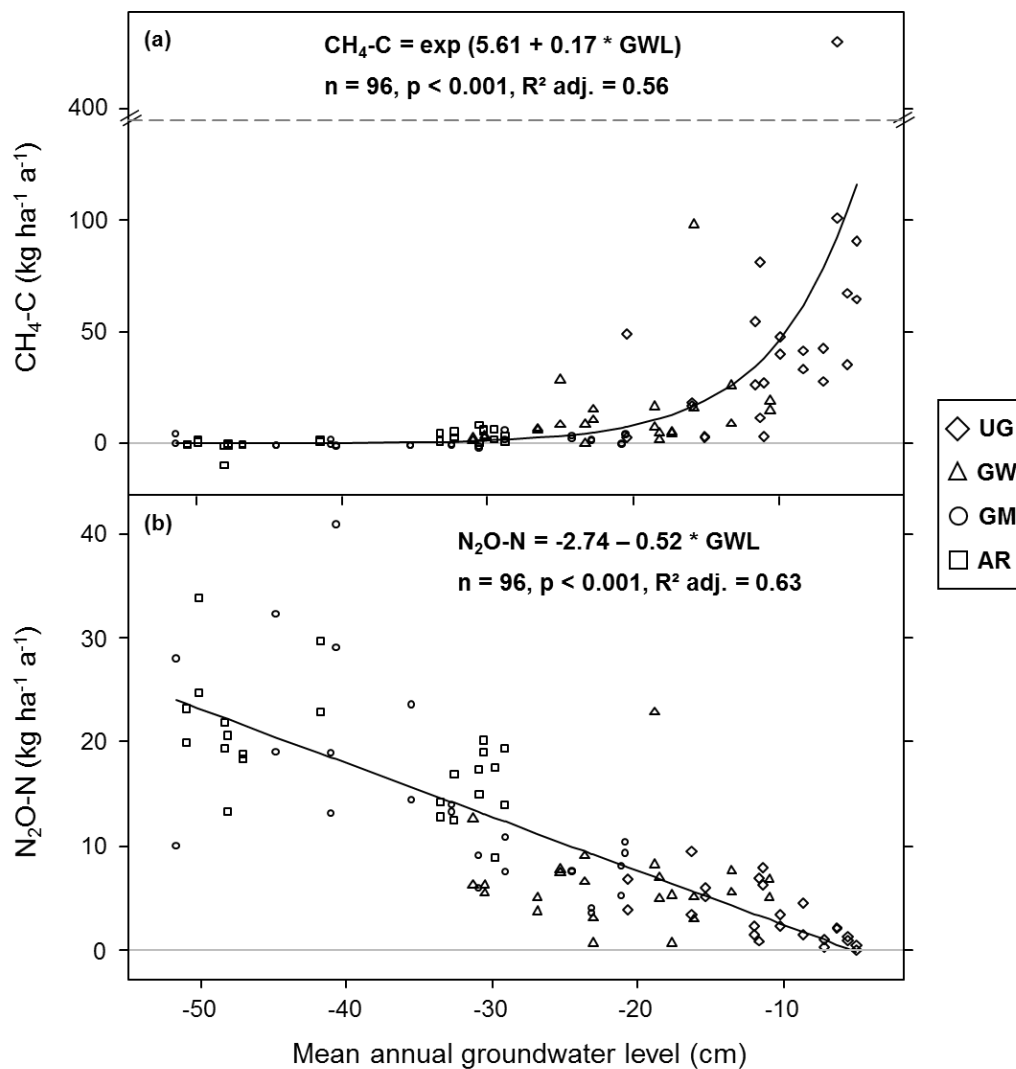
5 R<sup>2</sup> adjusted was estimated for predicted versus obtained values.



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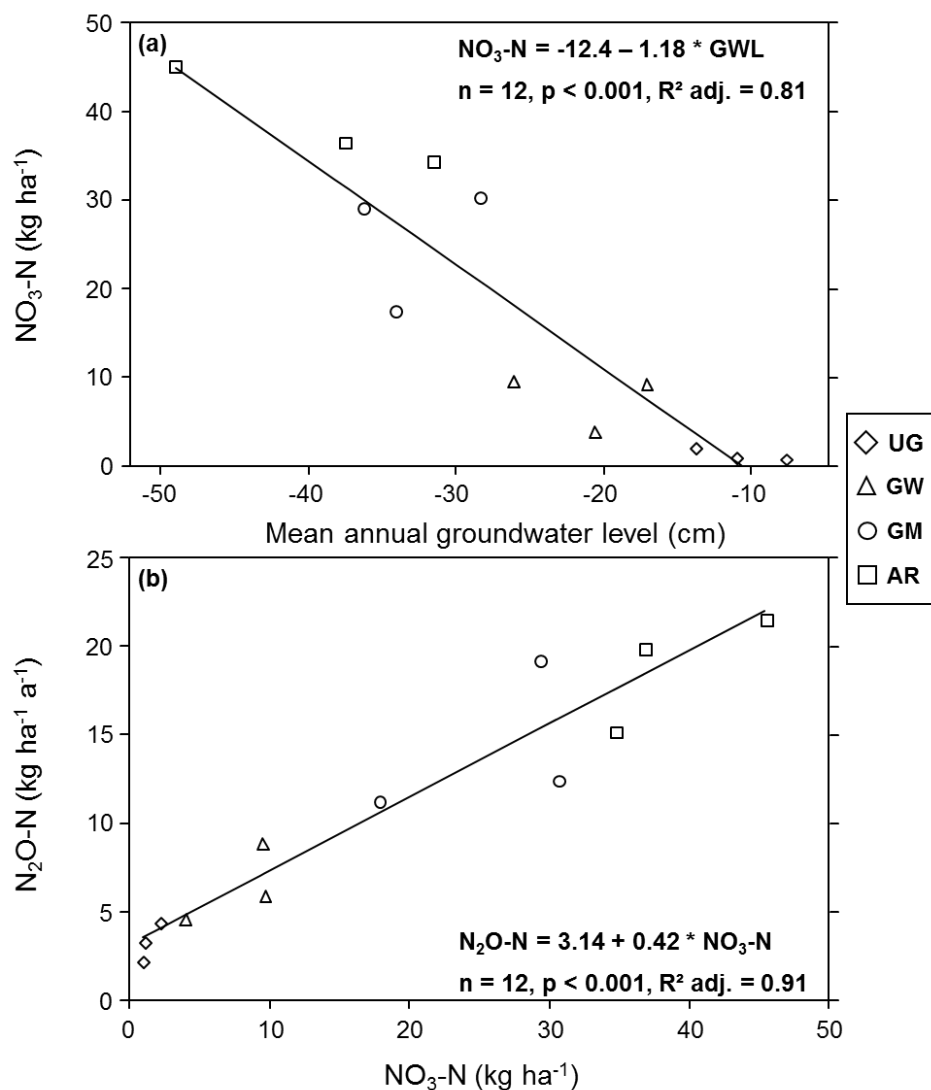
3 Figure 9. Relationship of cumulated N<sub>2</sub>O fluxes (n = 8) for a certain period of the growing seasons 2012  
4 and 2013 at the arable site (AR) to nitrogen balance (n = 3) for the same period, calculated from mineral  
5 N input of mineral and organic fertilizers and the N removal by plants. R<sup>2</sup> adjusted was estimated for  
6 predicted versus obtained values.



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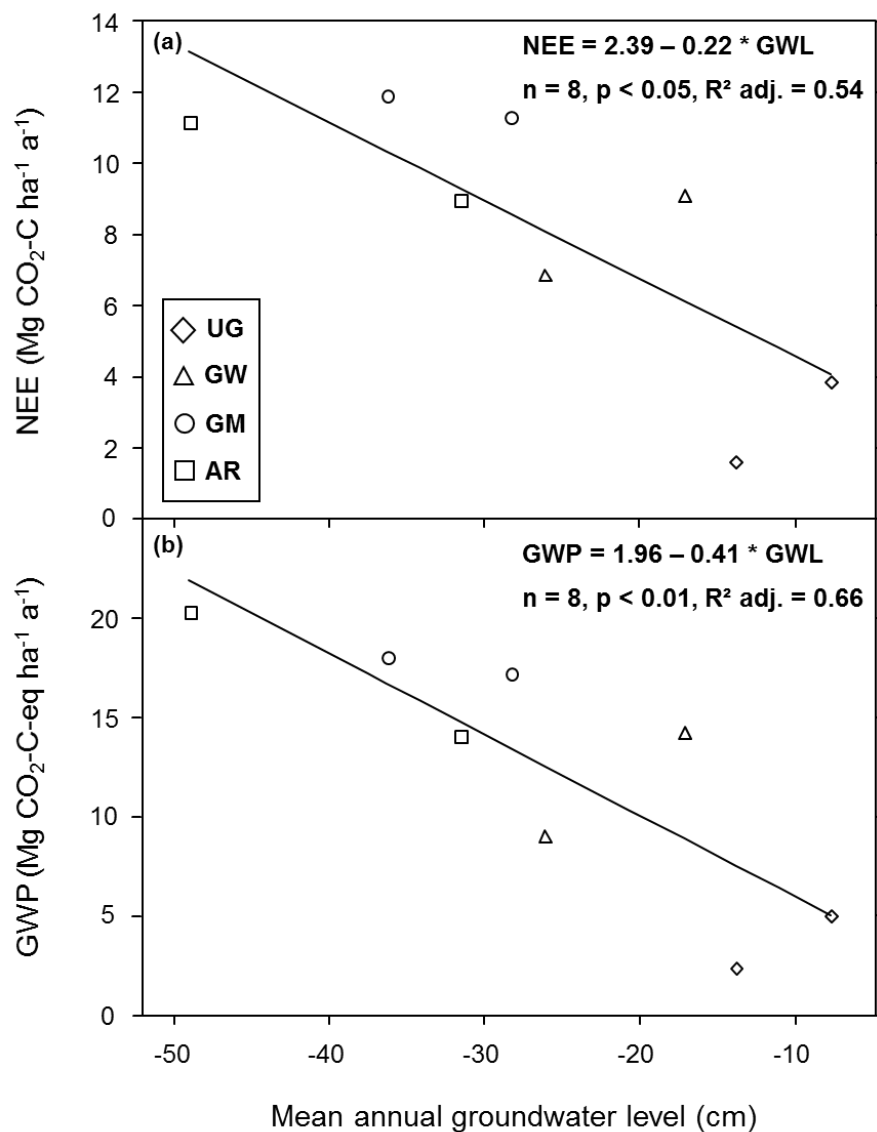
3 Figure 10. Relationships of cumulated annual CH<sub>4</sub> (a) and N<sub>2</sub>O fluxes (b) to mean annual groundwater  
 4 level for the study period (April 2011 – March 2014) with n = 8 per site and year. GWL in the equations  
 5 is mean annual groundwater level (cm). R<sup>2</sup> adjusted for exponential regression (a) was estimated for  
 6 predicted versus obtained values. Note the broken y-axis for figure (a).



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3 Figure 11. Relationship of mean annual amount of nitrate in 0 – 20 cm soil depth to mean annual  
 4 groundwater level (a) and relationship of mean annual  $\text{N}_2\text{O}$  balances to mean annual amount of nitrate  
 5 in 0 – 20 cm soil depth (b). GWL in the equation (a) is mean annual groundwater level (cm),  $\text{NO}_3\text{-N}$  in  
 6 (b) is mean annual amount of nitrate in 0 – 20 cm ( $\text{kg N ha}^{-1}$ ).



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3 Figure 12. Effect of mean annual groundwater level on net ecosystem exchange (NEE) of CO<sub>2</sub> (a) and

4 global warming potential (GWP) of the four study sites (b) during the period April 2012 – March 2014.

5 GWL in the equations is mean annual groundwater level (cm).