# Response to the reviews

- **How long the oven drying of soil was done to estimate the gravimetric water content? It usually takes 24 hours at 105 °C, but it is better to mention the duration.**

The soil samples were dried at 105 °C for roughly 24 hours. At the beginning of our study we tested if the sample weight would be further reduced during a second day of drying but we found that this was not the case.

- **How did you calculate organic C content from the elemental analyzer? The combustion method used in the elemental analyzer usually gives an estimate of total C. You can do acid digestion prior to the combustion step in order to eliminate the inorganic C or offline calculation can be done for organic C. If the assumption is that the peatland is mostly organic soil, then the estimates of loss on ignition are warranted. Please clarify this issue. Also, mention the time taken for oven drying of the samples at 40 °C.**

As the peat soil in our study was free of inorganic C, the total C determined by the combustion method equals the organic C content of the soil sample. Oven drying at 40 °C was done for 2 – 4 days, depending on the moisture content of the samples.

- **Is it normal to collect flux measurements between 9am to noon in these areas? When do you expect to see the peak in the diel pattern of $CO_2$? Peak in the $CO_2$ flux often lags by few hours with respect to the peak in the soil temperature in temperate and boreal forests (see Gaumont-Guay et al., 2006 and Savage et al., 2009) due to a delayed response to the aboveground processes. The time lag in the agricultural system may be much less, if any. Also, to capture the daily mean value, you should take representative readings before and after the flux value peaked. An explanation on this may be of worth to support the sampling time used in this study for a representative mean daily flux.**

During the mentioned period of the day we expected the mean daily fluxes of $N_2O$ and $CH_4$ as it was described in other studies. For example, van der Weerden et al. (2013) conducted near-continuous measurements of $N_2O$ fluxes with automatic chambers and stated that mean daily fluxes occurred between 10:00 and 12:00 h and 18:00 and 21:00 h.

We expected the peak GPP at the same time when PAR is at its maximum, which is generally between 12:00 and 13:00 h. Maximum daily $R_{ECO}$ is usually measured few hours later due to the time lag of temperature maxima. As $R_{ECO}$ is the sum of autotrophic and heterotrophic respiration, it depends on both, air and soil temperature. We did not expect a time lag between

the maxima of these temperatures and the respiration. However, our $CO_2$ measurement campaigns were conducted from sunrise until the afternoon (approximately 4 h after PAR peaked) to cover the daily range of radiation and temperature and thus assimilation and respiration. We usually stopped our daily campaigns when we saw no increase in the $R_{ECO}$ flux anymore.

- **What were your criteria for the acceptance of the $CO_2$ flux data? Did you follow the same approach like $CH_4$ and $N_2O$ flux data for the coefficient of determination?**

Quality criteria for $CO_2$ measurements were changes of chamber temperature of more than 1.5 °C and a standard deviation of PAR more than 10 % of average PAR for NEE measurements (transparent chamber). If these thresholds were exceeded during a measurement, the $CO_2$ flux was not used for further analyses. Additionally, each single $CO_2$ measurement was carefully checked and the flux was only calculated for that part of the measurement with a linear concentration change over time. The $R^2$ was not used as a quality criteria for the $CO_2$ fluxes as there were up to 60 data points ($CO_2$ concentrations) during one measurement and especially for measurements in winter times when fluxes were very low, a $R^2$ of ≥ 0.9 could hardly be fulfilled.

- **What do you mean by "own examination"? Please explain briefly.**

I measured the PAR inside and outside the chamber at different light intensities and found that PAR inside the chamber was on average 8 % lower than outside the chamber.

- **The equation for NEE is little confusing for the general reader. I see that you have mentioned the sign convention of individual flux components in L 19-21. But, it is better to write NEE=GPP - R_ECO and rephrase in the previous line that NEE was calculated as the difference (not sum) between GPP and R_ECO. It is better to maintain the conventional sign of flux: positive flux as a source to the atmosphere (which R_ECO is) and negative flux as a sink to the ecosystem (which GPP is) and the net balance of these two ultimately determine whether the ecosystem serves as a source (positive NEE) or a sink (negative NEE) for $CO_2$.**

We applied the conventional sign convention just like you mentioned it. As GPP is negative, $R_{ECO}$ has to be added to get the NEE. In recent studies observing the NEE of peatland ecosystems, the equation is written as a sum of the two processes and we wanted to be consistent at this point.

- **It will be worth exploring if the Kolmogorov-Smirnov Goodness-of-Fit Test for normality of data corroborates with the graphical residual analysis, especially for $CH_4$ and $N_2O$, which are often characterized by hot-spots or hot moments.**

As we apply the graphical residual analysis as a standard procedure to all data sets as a decision support tool for the statistical analyses, it was done in the same way with the $CH_4$ and $N_2O$ flux data. Pre-tests are often not recommended any more (see Rasch et al., 2011). An important reason is that pre-tests such as the Kolmogorov-Smirnov-Test only indicate if the data are not significantly normal distributed but they cannot give evidence about normal distribution which we would need to know. Therefore, we used the graphical residual analysis to determine the distribution of data.

- **In general, fluxes of $N_2O$ (and $CO_2$) have been reported in literature after thawing of frozen soil due to the release of stored labile C and nutrients. The buildup of these labile substrates during freezing event usually comprised of dead microorganisms, dead fine roots, and C released from the breakdown of aggregates. Also, the response often depends on the intensity and duration of freezing as well as the soil properties. So, please explain clearly your point on the pulse of $N_2O$ during the freezing event, in addition to the thawing event afterward? Also, it has been reported that the successive pulse of $N_2O$ has been reduced with increased frequency of freeze-thaw cycle, which may explain the lower winter fluxes in the second year. See Xu et al. (2016) and the articles cited in the reference list for more details.**

Our results show that the mentioned $N_2O$ pulse is occurring during freezing events but $N_2O$ fluxes decline rapidly after freezing. This was more pronounced when no snow cover was present. These two points suggest that the predominating process that enhanced winter $N_2O$ fluxes was the freezing rather than the thawing of the peat soils. As the $N_2O$ flux did not increase directly after air temperatures became negative but a few days afterwards, underlines this conclusion as the wet peat soils have a high heat capacity, which means that the time lag between changes in air and soil temperatures is relatively great. This could explain the missing $N_2O$ pulse in the second winter as the frost could not penetrate the peat sufficiently to generate an enhanced release of C and N. However, due to the very high amounts of C stored in the peat soils and the densely rooted top soils in combination with high nutrient loads on the agricultural sites, the investigated peatlands have a high potential for $N_2O$ emissions in general and also for freezing induced $N_2O$ pulses.

- **Please note the prerequisite for the release of $N_2O$ in the incomplete denitrification process (where, complete denitrification: $NO_3^-$ -> $NO_2^-$ -> NO -> $N_2O$ -> $N_2$) is the**

**onset of anoxic (or reduced) condition. Do you have evidence that the $N_2O$ emission was greatest from nitrate-rich soils (or soil microsite) with relatively greater water filled pore space?**

$N_2O$ emissions were not significantly related to water filled pore space. The ground water level (GWL) was the dominating factor for annual $N_2O$ emissions at our sites with increasing emissions at lower mean annual GWL. However, it has to be noted that with lower mean annual GWL also the fluctuations of the GWL are increasing, which means that there is a thicker active peat layer where N can be mineralized and nitrified. The produced nitrate that is dislocated to saturated pores will then be denitrified with potential losses of $N_2O$. As a consequence, daily $N_2O$ fluxes of peat soils can hardly be related to the GWL or water filled pore space at a certain day, at least in field studies. We only found a significant relation between the daily fluxes and the amount of nitrate in the topsoil as the occurrence of high amounts of nitrate that exceed plant uptake can lead to incomplete denitrification and thus $N_2O$ release.

- **$R^2$ adj for the model is 0.05. Does this mean ground water level and soil temperature at 5 cm depth could explain only 5% of the variation in the flux of $CH_4$-C? Please clarify the relevance of the model and what is the interpretation of this figure**.

The figure is not an illustration of the model as we used a multiple linear regression model with log-transformed daily $CH_4$ fluxes where the site was used as a covariate, additionally to the groundwater level and soil temperature at 5 cm depth. The figure was made to illustrate the extremely high variability of $CH_4$ fluxes between the sites but also within single sites and to show that highest emissions occurred when both groundwater level and soil temperature were high. However, this was highly depending on the location as the deep-drained sites showed negligible fluxes irrespective of GWL and soil temperature. This was underlined by the model as all three covariates had a highly significant effect on $CH_4$ fluxes. In addition, the model comprised additional terms as for example the year as a random factor and a heteroscedasticity term due to site and year. However, this model only explained 11 % of the variation in the $CH_4$ flux. The $R^2$ adj. in Fig. 8 indicates that only 5 % are explained when only the GWL and the soil temperature are considered as influencing factors.

# List of changes

- **Sec. 2.2.2:** The duration of oven drying at 105 °C (p10, l 23) and at 40 °C (p10, l 25) as well as a note on the determination of organic C content (L 26 – 28) has been added.
- **Sec. 2.3.1:** Notes on the time of gas sampling (p 11, l 28 – 29) and $CO_2$ measurements (p 12, l 18 – 20) have been added.
  The last sentence has been removed (p 12, l 21 – 23).
- **Sec. 2.3.2:** The explanation of quality criteria for $CO_2$ flux measurements has been added (p 13, l 14 -16).
- **Sec. 2.3.3:** The determination of PAR absorption by transparent chambers is explained in more detail (p 14, l 19 – 21).
- **Sec. 2.4:** A reference has been added (p 15, l 24 – 25).
- **Sec. 4.1:** A note on the $CH_4$ model's explanatory power was added, including a citation (p 20, l 27 – 29).
- **Sec. 4.2:** A note on the $N_2O$ model's explanatory power was added, including a citation (p 22, l 12 – 14).
  $N_2O$ fluxes during freeze-thaw events were discussed in more detail (p 23, l 31 – p 24, l 10).
- **References:** Additional references were added:
  Nakagawa & Schielzeth (2013) (p 39, l 1 – 3)
  Rasch et al. (2011) (p 39, l 26 – 27)
  Xu et al. (2016) (p 43, 4 – 6)

# Greenhouse gas emissions from fen soils used for forage production in northern Germany

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## Abstract

A large share of peatlands in northwest Germany is drained for agricultural purposes, thereby emitting high amounts of greenhouse gases (GHG). In order to quantify the climatic impact of fen soils in dairy farming systems of northern Germany, GHG exchange and forage yield were determined on four experimental sites which differed in terms of management and drainage intensity: a) rewetted and unutilized grassland (UG), b) intensive and 'wet' grassland (GW), c) intensive and 'moist' grassland (GM) and d) arable forage cropping (AR). Net ecosystem exchange (NEE) of $CO_2$ and fluxes of $CH_4$ and $N_2O$ were measured using closed manual chambers. $CH_4$ fluxes were significantly affected by groundwater level (GWL) and soil temperature, whereas $N_2O$ fluxes showed a significant relation to the amount of nitrate in top soil. Annual balances of all three gases, as well as the global warming potential (GWP), were significantly correlated to mean annual GWL. Two-year mean GWP, combined from $CO_2$-C-equivalents of NEE, $CH_4$ and $N_2O$ emissions, as well as C input (slurry) and C output (harvest), was 3.8, 11.7, 17.7 and 17.3 Mg $CO_2$-C-eq $ha^{-1}$ $a^{-1}$ for sites UG, GW, GM and AR, respectively (standard error (SE) 2.8, 1.2, 1.8, 2.6). Yield related emissions for the three agricultural sites were 201, 248 and 269 kg $CO_2$-C-eq (GJ net energy lactation (NEL))$^{-1}$ for sites GW, GM and AR, respectively (SE 17, 9, 19). The carbon footprint of agricultural commodities grown on fen soils depended on long-term drainage intensity rather than type of management, but management and climate strongly influenced interannual on-site variability. However, arable forage production revealed a high uncertainty of yield and therefore was an unsuitable land use option. Lowest yield related GHG emissions were achieved by a three-cut system of productive grassland swards in combination with a high GWL (long-term mean ≤ 20 cm below the surface).

# 1 Introduction

Natural peatland ecosystems act as long-term carbon (C) sinks as C in plant residues accumulates due to anoxic conditions and thus incomplete decomposition (Joosten & Clarke, 2002). Globally, the amount of C stored in peatlands is about 446 Pg (2 Pg in German peatlands) (Joosten, 2009), which is 24 % higher compared to the number of 359 Pg C stored in global forest vegetation, given by Dixon et al. (1994). The drainage of peatlands causes aerobic soil conditions, leading to accelerated mineralization of the soil organic matter (SOM) and an increased release of C and nitrogen (N) (Höper, 2002). Therefore, the natural sink for C and N is turned into a net source, converting drained peatlands to significant emitters of the greenhouse gases carbon dioxide ($CO_2$) and nitrous oxide ($N_2O$) (Kasimir-Klemedtsson et al., 1997; Maljanen et al., 2003b, 2010). Simultaneously, the methane ($CH_4$) emissions occurring under natural conditions are reduced to negligible levels (Roulet et al., 1993; van den Pol-van Dasselaar et al., 1997; Maljanen et al., 2003a).

In Germany, peatlands cover around 1.67 million ha (Joosten, 2009), which corresponds to 4.7 % of the land area. Roughly 65 % of these peatlands are minerotrophic fens (Grosse-Brauckmann, 1997) and around 70 % is utilized for agricultural purposes (Röder & Osterburg, 2012). Peatland rich regions, as particularly northwest (NW) Germany (Lower Saxony, Schleswig-Holstein), show high shares of forage production and livestock units per ha of utilized agricultural area, which is attributed to a concentration of dairy farming (Röder & Osterburg, 2012). Consequently, there is a high demand for intensive forage production to ensure the supply of a high quality fodder. These management and cultivation practices require an intensive drainage and fertilization, leading to a continually increasing pressure on the utilization of German peatlands. The relevance of agriculturally utilized peatlands for the national GHG budget is highlighted as only 5 % of the utilized agricultural area (Röder et al., 2011) but 50 % of the GHG emissions from agricultural soils (41.3 of 82.7 Tg $CO_2$-equivalents ($CO_2$-eq)) are attributed to peatlands drained for agriculture (UBA, 2014).

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

protection measures for these areas should focus on resource use efficiency, i.e. minimizing GHG emissions per unit of product instead of unit area (Oenema et al., 2014). Here, we will focus on the net exchange of the three biogenic trace gases $CO_2$, $CH_4$ and $N_2O$ from fen soils in an intensive dairy farming region of northern Germany (Schleswig-Holstein) and relate their annual budgets to forage energy yield (net energy lactation, NEL) of the specific sites.

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

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

(i) rewetting leads to a decrease in $CO_2$ and $N_2O$ emissions but an increase in $CH_4$ emissions,

(ii) the GHG balances and C losses increase with land use intensity in the order UG > GW > GM > AR,

(iii) product related GHG emissions are higher for arable forage cropping on organic soils compared to grassland utilization,

(iv) wet but intensive grassland utilization (site GW) realizes lowest product related GHG emissions.

# 2 Material and Methods

## 2.1 Study area

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

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

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

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

## 2.2 Site characteristics

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

### 2.2.1 Groundwater levels

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

### 2.2.2 Soil properties

For monitoring of soil mineral N status, soil samples were taken fortnightly with a soil auger at a depth of 0 – 20 cm on each site. Nitrogen was extracted with 0.01 M $CaCl_2$ (VDLUFA, 1997) and the concentrations of nitrate ($NO_3^-$) and ammonium ($NH_4^+$) of the extractions were analyzed photometrically with a dual channel continuous flow analyzer (San$^{++}$, Skalar Analytical B.V., Breda, The Netherlands). Mineral N stocks per ha were calculated using the bulk density of the relevant sites. Bulk density was determined for the depths 5, 15, 25 and 45 cm according to DIN ISO 11272 (HBU, 1998). The gravimetric water content of soil samples was estimated by oven drying at 105 °C for 24 h. To calculate the contents and amounts of $C_{org}$ and $N_{tot}$ of each site, soil sampling was conducted twice a year at soil depths of 0 – 30, 30 – 60 and 60 – 90 cm. After oven drying (40 °C for 48 – 96 h), samples were analyzed with an elemental analyzer (Vario Max CN, Elementar, Hanau, Germany). As the peat soils were free of inorganic C, the total C determined by the combustion method equaled the organic C content. The soil pH was determined before and after the study period in 2011 and 2014 according to VDLUFA (1991).

### 2.2.3 Herbage yield and forage quality

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

## 2.3 Determination of GHG fluxes and balances

### 2.3.1 Flux measurements

$CH_4$ and $N_2O$ fluxes were measured from April 2011 to March 2014 using closed manual chambers (Hutchinson & Mosier, 1981). Measurements were conducted weekly and in addition shortly after management practices like fertilization or tillage. At each site, eight PVC collars (d = 60 cm, h = 15 cm) were inserted 10 cm into the soil one week before the measurements started. To display gas fluxes for different GWLs at the same time, four collars were placed at 5 and 15 m from the next drainage ditch, respectively. When sites were harvested, the vegetation was removed from the collars. Site preparation measures were conducted in spring and the collars were shifted afterwards to obtain representative conditions. On site UG, a boardwalk was installed due to wet soil conditions and to avoid disturbances around the collars. For gas flux measurements, opaque PVC chambers (h = 35 cm, V = 0.1 $m^3$) were used and chamber air samples were collected with a 30 ml syringe and stored in 12 ml pre-evacuated septum capped vials (Labco, High Wycombe, UK) (Glatzel & Well, 2008) 0, 15 and 30 min after chamber closure. Sampling was conducted between 09:00 and 12:00 h as it has been shown that mean daily fluxes generally occur during this period of the day~~to capture mean daily fluxes~~ (Velthof & Oenema, 1995a; Petersen et al., 2012; van der Weerden et al., 2013). The samples were analyzed for concentrations of $CH_4$, $N_2O$ and $CO_2$ with a gas chromatograph (7890a, Agilent Technology Inc., Santa Clara, CA, USA) equipped with a flame ionization detector (FID), electron capture detector (ECD) and

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

The $CO_2$ exchange was determined according to the method of Drösler (2005). Elsgaard et al. (2012), Beetz et al. (2013) and Leiber-Sauheitl et al. (2014) present similar approaches. Here, static chambers with a diameter of 61 cm and a height of 35 were used. On each site three PVC collars were installed. Measurement campaigns were conducted during the period March 2012 until April 2014 in intervals of 3 to 5 weeks. When harvest of the agricultural sites took place, the vegetation was removed from the collars and additional $CO_2$ measurements were carried out few days after harvest. In total, the $CO_2$ exchange was measured on 21, 28, 30 and 32 days at site UG, GW, GM and AR, respectively. Transparent and opaque chambers were used to measure the net ecosystem exchange (NEE) and the ecosystem respiration ($R_{ECO}$), respectively. The chambers were connected to an infrared gas analyzer (LI-820, LI-COR Biosciences, Lincoln, NE, USA) and a data logger (CR 1000, Campbell Scientific, Logan, UT, USA). $CO_2$ concentration inside the chamber, temperature inside and outside the chamber and PAR outside the chamber were recorded every 5 s. Chambers were equipped with a fan to ensure homogenization of the atmosphere inside the chamber headspace. When the vegetation was higher than the chambers, extensions (h = 35 cm) were used. Due to the time lag between the maxima of PAR and temperature (air and soil), ~~M~~measurement campaigns were conducted from sunrise until afternoon to comprise the whole daily range of the driver variables~~PAR and soil temperature~~. Maximum enclosure times were 120 s for NEE and 300 s for $R_{ECO}$ measurements. ~~Quality criteria for $CO_2$ measurements were changes of chamber temperature of more than 1.5 °C and a standard deviation of PAR more than 10 % of average PAR. Measurements that exceeded these threshold values were discarded.~~

### 2.3.2 Flux calculations

Trace gas fluxes were calculated using linear regression for the change of gas concentration over time as it has been described in several other studies (e.g. Flessa et al., 1998; Chatskikh et al., 2008; Beetz et al., 2013). Since effects of temperature and pressure inside the chamber induce only minor uncertainties to the measured fluxes (Levy et al., 2011), these variables are often neglected in flux calculations (Chatskikh et al., 2008). However, to quantify the uncertainty in calculated $CO_2$ fluxes caused by a varying density of air as a function of temperature, $CO_2$ fluxes (n = 5546) were corrected for the mean temperature inside the chamber and compared to the uncorrected fluxes. On average, temperature correction reduced calculated fluxes by 6 % with a maximum reduction of 12 % at a very high temperature of 38 °C. As temperature was not measured inside the chambers for $CH_4$ and $N_2O$ flux

measurements, the uncorrected $CO_2$ fluxes were used for further analyses to ensure methodological consistency.

For $CH_4$ and $N_2O$, fluxes were accepted when the coefficient of determination ($R^2$) of the linear regression was ≥ 0.9 to ensure a high accuracy of measured fluxes. Measurements with $R^2 < 0.9$ occurred mainly when chamber concentrations were near ambient and the corresponding fluxes were assumed to be 0. $CO_2$ concentrations of the gas samples were used as control to identify erroneous $CH_4$ and $N_2O$ values. If the $CO_2$ concentration of a sample was not plausible (i.e. smaller than previous), the fluxes of $CH_4$ and $N_2O$ were discarded from the dataset (Leiber-Sauheitl et al. 2014). For NEE and $R_{ECO}$ measurements, all fluxes with plausible concentration changes over time were accepted, irrespective of flux magnitude and the $R^2$ of linear regression (Alm et al., 2007b; Leiber-Sauheitl et al., 2014). To avoid underestimation of $CO_2$ exchange by a diminishing concentration gradient between chamber headspace and soil or plant, and thus decreasing fluxes (Davidson et al., 2002), only the part of linear concentration change was used for flux calculation, which could be only 30 s for NEE measurements with highly productive vegetation and high PAR. Quality criteria for $CO_2$ measurements were changes of chamber temperature by more than 1.5 °C and a standard deviation of PAR more than 10 % of average PAR. Measurements that exceeded these threshold values were discarded.

### 2.3.3 $CO_2$ modelling

$R_{ECO}$ was estimated using a temperature-dependent flux model according to Lloyd & Taylor (1994):

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

where $R_{ECO}$ is the measured ecosystem respiration (g $CO_2$-C $m^{-2}$ $h^{-1}$), $R_{ref}$ is the respiration at reference temperature (g $CO_2$-C $m^{-2}$ $h^{-1}$), $E_0$ is an activation-like parameter (K), $T_{ref}$ is the reference temperature (283.15 K), $T_0$ is the temperature constant for the start of biological processes (227.13 K), and T is the temperature with the best fit to the data of one measurement campaign. This could be either soil temperature in 5 cm depth at the corresponding site or the air temperature from the weather station at site GW. For modelling $R_{ECO}$, $R_{ref}$ and $E_0$ were fitted plot based for each measurement campaign with soil or air temperature, depending on the level of significance. If neither soil temperature nor air temperature gave a significant relation to $R_{ECO}$ of a measurement campaign, the data was pooled with that of one or two adjacent campaigns to obtain significant parameters for the $R_{ECO}$ model (Beetz et al., 2013). However, for site UG it was in some cases not possible to calculate significant parameters. Therefore, the dataset was separated into growing season and non-growing season according to Janssens (2010) and all measurement campaigns of a season were pooled. By this approach, the temporal

resolution of the model was decreased, but the range of temperatures for which the model is valid, was greatly increased. Nevertheless, for the agricultural sites it was necessary to consider the phenological development of the plants and especially the effect of harvest in the model. When fitting the model per campaign, the temperature range can be very narrow, which may lead to severe overestimations by the $R_{ECO}$ model if the slope of regression is high and the temperature is above of the observed range. Therefore, the highest measured value of the corresponding campaign was set as a threshold for maximum $R_{ECO}$. Every modelled value exceeding that threshold was recessed. The fitted parameters $R_{ref}$ and $E_0$ were linear interpolated between the campaigns and $R_{ECO}$ was modelled on an hourly basis using the corresponding temperature. To calculate GPP, the modelled $R_{ECO}$ at the time of NEE measurements was subtracted from the measured NEE value.

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

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

where GPP is the calculated gross primary production (g $CO_2$-C $m^{-2}$ $h^{-1}$), $GP_{max}$ is the limit of carbon fixation for infinite PAR (g $CO_2$-C $m^{-2}$ $h^{-1}$), α is the initial slope of the regression curve or light use efficiency ((g $CO_2$-C $m^{-2}$ $h^{-1}$) (µmol $m^{-2}$ $s^{-1}$)$^{-1}$) and PAR is the average photon flux density of photosynthetically active radiation (µmol $m^{-2}$ $s^{-1}$) that was determined during the NEE measurement by a quantum sensor (SKP 215, Skye Instruments, Llandrindod Wells, UK). PAR was corrected by a factor of 0.92 as a~~n~~ mean absorption by the transparent chambers of 8 % ~~by the transparent chambers~~ was identified by measuring the PAR inside and outside the chambers at different light intensities~~own examinations~~. $GP_{max}$ and α were fitted plot based for each measurement campaign and linear interpolated between the campaigns, assuming a consistent development of vegetation. However, as the plant biomass is harvested, $CO_2$ uptake is interrupted immediately. Therefore, the parameters of the preceding measurement campaign, which was conducted only few days before harvest, were used until the cutting and then set back to 0. The subsequent campaign was conducted within one week after the cutting to capture the $CO_2$ exchange of the recently harvested plants. GPP was modelled on an hourly basis using measured PAR from the weather station at site GW.

### 2.3.4 GHG and C balances

As the net ecosystem exchange (NEE) of $CO_2$ is the balance of $CO_2$ uptake by plants (GPP) and the autotrophic and heterotrophic respiration of plants and soil ($R_{ECO}$) (Chapin et al., 2006), NEE was calculated on an hourly basis as the sum of Eqs. (1) and (2):

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

For further processing, GPP, $R_{ECO}$ and NEE were calculated per hectare and summed up to daily values (kg $CO_2$-C $ha^{-1}$ $d^{-1}$). The site specific annual balances of the three components were calculated as the average of the 365-days sums of the three replicates. Annual $CH_4$ and $N_2O$ balances were determined by plot based linear interpolation between the measurement days and summation of daily values. Site specific balances were calculated as average of the eight replicates. The global warming potential (GWP) of a specific site indicates to which magnitude it contributes to global warming, based on the GHG balance for a certain period. GWP was calculated using the IPCC (2007) radiative forcing factors of the individual gases for a time horizon of 100 years. These are 25 for $CH_4$ and 298 for $N_2O$ related to $CO_2$ ($CO_2$-equivalents ($CO_2$-eq)). Additionally, anthropogenic C inputs and losses via slurry application and harvest were calculated as $CO_2$-eq and included in the GWP (Beetz et al., 2013). Using the balances of $CO_2$-C and $CH_4$-C as well as the C import via slurry and C export via biomass harvest, the net ecosystem carbon balance (NECB) was calculated per site and year. For all C and GHG fluxes and balances, the atmospheric sign convention was applied, where all losses from the atmosphere into the ecosystem (site) are displayed as negative (the ecosystem acts as a sink) and all enrichments in the atmosphere are displayed as positive (the ecosystem acts as a source). This convention is transferred to the non-atmospheric fluxes like slurry application (negative) and biomass harvest (positive). GHG and carbon balances were calculated for the periods April 2012 – March 2013 and April 2013 – March 2014.

## 2.4 Statistical analyses

The statistical software R (2014) was used to evaluate the data. Evaluation started with the definition of an appropriate statistical mixed model (Laird & Ware, 1982; Verbeke & Molenberghs, 2000). The data were assumed to be normally distributed and heteroscedastic due to the different sites and measurement periods. These assumptions ~~a~~were based on a graphical residual analysis which was preferred to the application of pre-tests (Rasch et al., 2011). The statistical model included the site as a fixed factor. For daily $CH_4$ fluxes, GWL and soil temperature in 5 cm were modelled as covariates, whereas for $N_2O$ fluxes, the amount of nitrate in 0 – 20 cm soil depth was used. The year was regarded as a random factor. Also, the correlations of the measurement values due to the day of sampling were taken into account. Based on this model, an analysis of covariance (ANCOVA) was conducted to test for significant influences of the covariates.

For balances of $CH_4$, $N_2O$ and $CO_2$, as well as for the GWP, NECB and product related GHG emissions, a mixed model with the site as fixed factor and the year as random factor was defined in each case. Heteroscedasticity was modelled due to the different sites and measurement periods. An analysis

of variance (ANOVA) was conducted to identify significant differences between the sites. For the yields of DM, C, N and NEL, the model was amplified by the year as a fixed factor instead of random factor. Furthermore, multiple contrast tests (Bretz et al., 2011) were conducted in order to identify significant differences between sites and years, respectively.

To evaluate the influence of GWL on the different trace gas balances and the total GWP, NECB and product related GHG emissions, mean annual GWL was added as a fixed factor to the model used for the *t*-test. This model was calculated with and without the interaction term of site and GWL, as well as irrespective of the different sites. These three model types were compared referring to their Akaike information criterion (AIC) (Akaike, 1974) to assess which model gives the best estimate for the relation between GWL and the corresponding variable. For $CH_4$ and $N_2O$ balances, this procedure was conducted for mean annual GWL and, in terms of $N_2O$, for mean annual soil nitrate.

For uncertainty analysis of the $CO_2$ model, a Monte Carlo simulation was conducted for each measurement plot and site. Therefore, model parameters with the same variation as the original values were randomly calculated for every measurement campaign or pooled dataset and new regressions with temperature ($R_{ECO}$) and PAR (GPP) were fitted. Only regressions with realistic parameters were accepted ($E_0$ and $R_{ref} \geq 0$, $\alpha$ and $GP_{max} \leq 0$). This procedure was conducted 10,000 times, thus, 10,000 different model outputs for $R_{ECO}$, GPP and NEE were obtained. The variation of these randomly calculated model outputs represents the uncertainty that is caused by the chamber measurements and by the fitting and linear interpolation of different numbers of measurement campaigns per plot and year. Since this procedure is conducted for each plot, the uncertainty can be calculated as the sum of mean variance of the three plots per site and the variance resulting from averaging the three replicates. This uncertainty was used for comparison of means obtained by the original simulation. Leiber-Sauheitl et al. (2014) present a similar approach.

# 3 Results

## 3.1 Weather conditions

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

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

## 3.2 Groundwater levels

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

## 3.3 GHG fluxes

### 3.3.1 Methane ($CH_4$)

Daily methane fluxes were highest at site UG and low at the agricultural sites (Fig. 3). While the intensively drained sites GM and AR showed negligible $CH_4$ exchange, $CH_4$ fluxes were on a higher level at site GW with one distinct emission peak in April 2013. $CH_4$ emissions from site UG showed high spatial and temporal variability. Emissions increased for the first time in August 2011, followed by a continuous release of $CH_4$ until July 2013. Highest emission peaks were recorded in summer 2012 and after that high releases occurred in autumn 2012 and spring 2013. Remarkably, the $CH_4$ flux pattern at site UG changed substantially in July 2013 as emissions ceased and did not rise again until the end of the study period in spring 2014.

### 3.3.2 Nitrous oxide ($N_2O$)

$N_2O$ fluxes during the 3-year period showed no distinct regularity at the unutilized site (UG), whereas the agricultural sites showed seasonal flux patterns with several emission peaks during spring, mainly

occurring after N fertilization (Fig. 4). While emissions at site UG peak in May 2013, the highest $N_2O$ releases from site GW were observed in April 2012. Similar but more frequent emission peaks were recorded at site GM in April and May 2012 and 2013 and further distinct $N_2O$ releases from that site were observed in autumn and winter 2013. The most pronounced seasonality of $N_2O$ emissions was determined at the arable site (AR) with high releases at the beginning of each study period. Thereby, the emissions in May 2013 clearly exceeded those of the preceding two years.

### 3.3.3 Carbon dioxide ($CO_2$)

The carbon dioxide exchange of the study sites was characterized by seasonal patterns of gross primary production (GPP) and ecosystem respiration ($R_{ECO}$) with high exchange rates during the vegetation period and smaller fluxes between October and April (Fig. 5). Maximum $CO_2$ uptake rates were -176, -188, -228 and -320 kg $CO_2$-C $ha^{-1}$ $d^{-1}$ for sites UG, GW, GM and AR, respectively (SE 9, 7, 17, 11). While this maximum C fixation took place in July 2013 at site UG, the two utilized grassland sites showed highest productivity in May 2012 before the first cutting. At site AR, maximum $CO_2$ uptake was modelled for the spring barley in June 2012. After the harvest of barley in August 2012, weeds remained that were eliminated by pesticides and mulched in September, so no $CO_2$ uptake could occur until emergence of newly seeded plants in May 2013. Maximum modelled $CO_2$ releases by $R_{ECO}$ from sites UG, GW, GM and AR were 156 (August 2012), 231 (May 2012), 216 (August 2012) and 259 kg $CO_2$-C $ha^{-1}$ $d^{-1}$ (June 2012), respectively (SE 16, 6, 2, 11). Depending on the extent of daily GPP and $R_{ECO}$ fluxes, the sites can act as net source or sink for $CO_2$. In total of two years (730 days), sites UG, GW, GM and AR acted as a $CO_2$ sink on 182, 156, 102 and 115 days, whereas they showed a net $CO_2$ release on 548, 574, 628 and 615 days, respectively (Fig. 5).

## 3.4 GHG balances, NECB and GWP

### 3.4.1 $CH_4$ and $N_2O$ balances

Over the three-year study period, mean annual $CH_4$ emissions were 55.1, 13.5, 0.9 and 1.8 kg $CH_4$-C $ha^{-1}$ $a^{-1}$ for sites UG, GW, GM and AR, respectively (SE 17.2, 4.0, 0.5, 0.7). Highest annual $CH_4$ release occurred at site UG in the second year, while minimum budgets were determined for sites GM and AR in the third year (Table 2). However, due to the low fluxes at sites GM and AR, cumulated annual $CH_4$ emissions were not significantly different from zero ($p > 0.05$). Sites GW and UG represented sources for $CH_4$ with significantly higher releases at site UG that also showed the highest variation in annual $CH_4$ budgets (Fig. 6a). Mean annual $N_2O$ balances of the four sites increased in the order UG, GW, GM and AR, accounting for 3.4, 6.5, 14.4 and 18.9 kg $N_2O$-N $ha^{-1}$ $a^{-1}$, respectively (SE 0.6, 0.9, 2.0, 1.1). Highest annual $N_2O$ emissions were recorded at site AR in the third year, whereas site UG released

minimum amounts of $N_2O$ in the second year (Table 2). The high budgets of sites GM and AR showed high variation and thus, did not differ significantly ($p = 0.18$) (Fig. 6b).

### 3.4.2 $CO_2$ balances and NECB

For the two years of $CO_2$ exchange measurement, mean annual NEE was 2.8, 8.0, 11.7 and 10.1 Mg $CO_2$-C $ha^{-1}$ $a^{-1}$ for sites UG, GW, GM and AR, respectively (SE 2.5, 0.7, 1.2, 1.9) (Fig. 7a). Thus, all sites showed higher annual $R_{ECO}$ than GPP sums, with highest $R_{ECO}$ at site AR and lowest $R_{ECO}$ at site UG, both for the period 2013/14 (Table 2). Highest annual GPP was determined at site AR for 2013/14, whereas site GM showed lowest GPP during the same period. As for $R_{ECO}$ and GPP, both highest and lowest NEE occurred in 2013/14 at sites GM and UG, respectively (Table 2). As indicated by NECB, all sites were net C sources during the study period with mean annual losses of 2.8, 10.6, 15.7 and 15.0 Mg C $ha^{-1}$ $a^{-1}$ at sites UG, GW, GM and AR, respectively (SE 2.6, 1.1, 1.4, 2.4) (Fig. 7b). Consistent with NEE, a higher range of NECB was assessed for the period 2013/14 with lowest C losses at site UG and highest losses at site AR (Table 2). The NEE and NECB of sites GW and AR did not differ significantly (Figs. 7a and b). However, mean NECB of site AR tended to be higher compared to site GW with $p = 0.07$.

### 3.4.3 GWP

The GWP combines the $CO_2$-C-eqs of NEE, $CH_4$ and $N_2O$ emissions, as well as the anthropogenic C balances from slurry applications and biomass removals. For the study periods 2012/13 and 2013/14, mean annual GWP was 3.8, 11.7, 17.7 and 17.3 Mg $CO_2$-C-eq $ha^{-1}$ $a^{-1}$ for sites UG, GW, GM and AR, respectively (SE 2.8, 1.2, 1.8, 2.6) (Fig. 7c). The lowest (site UG) as well as the highest GWPs (site AR) were observed for 2013/14 (Table 2). NEE dominated GWP at all sites with mean shares ranging from 59 % at site AR to 72 % at site UG. However, as no biomass removal occurred on site UG, this site also showed the highest shares of $CH_4$ and $N_2O$, with each gas accounting for 14 % of the GWP on average of the two years. The GWPs of the agricultural sites were considerably influenced by the C balances of slurry inputs and harvested biomass, which accounted for 21, 23 and 27 % at sites GW, GM and AR, respectively.

## 3.5 Crop yields and yield related GHG emissions

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

For the grassland sites, all yield parameters were higher in 2012 than in 2013 (Table 3). While this reduction was significant for site GM, site GW showed no significant differences between years. At the arable site, significantly higher yields were obtained by spring wheat with undersown grass in 2013

compared to spring barley in 2012. Site GM revealed significantly higher yields than site AR in 2012, while site GW did not differ to any other site in that year, except for N yield. In 2013, yields of sites GM and AR showed no significant differences, while site GW had significantly lower yields than the other two sites, except for the N yield of site AR and the NEL yield of site GM. On average, site GM showed the highest yields, while lowest yields were observed on site GW, except for N yield, which was lowest on site AR. However, only N yield of sites GM and AR differed significantly.

### 3.5.2 Yield related GHG emissions

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

## 4 Discussion

### 4.1 $CH_4$ fluxes and balances

Sites GM and AR showed negligible $CH_4$ fluxes and annual $CH_4$ budgets were not significantly different from zero. This is in accordance with other observations on intensively used peat soils that report low $CH_4$ emissions or even net uptake of $CH_4$ (Flessa et al., 1998; Maljanen et al., 2003a, 2004; Schäfer et al., 2012). The water table is the main controlling factor for $CH_4$ emissions from peat soils, particularly in absence of aerenchymus shunt species. A drainage depth of 20 – 30 cm is regarded as sufficient to inhibit the diffusion of high amounts of $CH_4$ into the atmosphere as $CH_4$ produced in the anoxic zone is oxidized by methanotrophs in the unsaturated zone (Couwenberg, 2009a; Schäfer et al., 2012). Accordingly, the low $CH_4$ fluxes at sites GM and AR can be explained by the high drainage intensity. However, a high GWL close to or above the soil surface did not enhance $CH_4$ production and emission at these sites (Fig. 8). A multiple linear regression model showed significant relations between log-transformed daily $CH_4$ fluxes and site ($p < 0.001$), GWL ($p < 0.001$) and soil temperature at 5 cm depth ($p < 0.01$). However, the model only explained 11 % of the variation in the $CH_4$ flux data (Nakagawa & Schielzeth, 2013) indicating the high complexity of $CH_4$ emission patterns and its relations to driver variables. Also,~~Therefore,~~ reactions on alterations of GWL and soil temperature differed between sites, probably as a consequence of long-term adaptation of methanogenic and methanotrophic communities to drainage intensity (van den Pol-van Dasselaar et al., 1997; Yrjälä et al., 2011). At site GW, $CH_4$ production potential was higher compared to sites GM and AR, leading to

considerable $CH_4$ releases, especially when GWL and soil temperature were high, as for example in summer 2012 (Fig. 3).

Conspicuous $CH_4$ peaks were detected at site UG in 2012 (Figs. 3 and 8) that were associated with high GWLs due to heavy rain fall in July and high soil temperatures due to a heat wave in late July and August (Figs. 1 and 2). These conditions likely favored a rapid expansion of the methanogenic community, more pronounced than in summer 2011 when GWLs were similarly high but temperatures were lower. Nykänen et al. (1998) reported that peat temperature controls $CH_4$ dynamics at high water tables, whereas the correlation is poor at low water tables. This is confirmed by the situation at site UG in summer 2013 when $CH_4$ emissions ceased as a consequence of low precipitation and water level drawdown in July and August, although soil temperatures were high. The subsequent GWL rise in autumn had no effect on $CH_4$ emissions, which remained low until the end of the study period. A possible explanation is that the methanogenic community was impaired by oxidative stress in summer (Görres et al., 2013) and did not recover due to low soil temperature when GWL rose (Bubier & Moore, 1994). Knorr et al. (2008) reported that $CH_4$ production in a fen soil was retarded by experimental drought for up to several weeks after rewetting. Estop-Aragonés & Blodau (2012) observed a longer time lag until $CH_4$ production recovered after rewetting for more intense and longer dried fen peat but warmer conditions favored the recovery. Furthermore, the dry soil conditions in summer 2013 could have increased the methanotrophic community, leading to a $CH_4$ consumption potential in the subsequent months exceeding the production potential as methanotrophic bacteria react less sensitively to temperature changes than methanogenic bacteria (Dunfield et al., 1993). This is supported by the results of this study as the overall highest daily $CH_4$ uptakes were measured at site UG in summer and autumn 2013.

Annual $CH_4$ balances of the study sites are comparable to those recently reported for temperate European peat soils (Schäfer et al., 2012; Beetz et al., 2013; Leiber-Sauheitl et al., 2014). Annual balances were significantly related to site and mean annual GWL (both with $p < 0.001$). Confirming the general understanding of $CH_4$ emission patterns (Couwenberg, 2009a), no significant $CH_4$ releases were observed for mean GWLs below -25 cm. At mean GWLs above -10 cm, $CH_4$ emissions were highly variable, with a minimum release of 28 and a maximum of 430 kg $CH_4$-C $ha^{-1}$ $a^{-1}$ (Fig. 10a), which is typical for the high spatial variability of $CH_4$ fluxes (Waddington & Roulet, 1996; van den Pol-van Dasselaar et al., 1999). The low contribution of $CH_4$ emissions to the GWP of the three agricultural sites (Table 2) illustrates the minor importance of $CH_4$ in terms of GHG mitigation on utilized peat soils. However, Hahn-Schöfl et al. (2011) showed that degraded fen grasslands can emit huge amounts of $CH_4$ as a consequence of flooding when easily degradable fresh plant material is present. Therefore, inundation of sites with highly productive, energy rich grasses such as perennial ryegrass (*Lolium*

*perenne*) bears the risk of enhanced $CH_4$ emissions, especially during summer. This should be particularly considered for site GW, where a significant $CH_4$ production potential could be observed.

### 4.2 $N_2O$ fluxes and balances

$N_2O$ emissions measured at the study sites were of similar magnitude as observed for other agricultural fen soils, for example in South Germany (Flessa et al., 1998), the Netherlands (van Beek et al., 2010; 2011) or Denmark (Petersen et al., 2012) and conform to the range of $N_2O$ hotspots on European organic soils given by Leppelt et al. (2014). The $N_2O$ release from site UG represents the emissions without agricultural utilization in the study area. These were higher than reported for natural peatlands (Leppelt et al., 2014), which might be a result of GWL fluctuations (Figs. 2 and 10b), as background $N_2O$ emissions strongly depend on drainage intensity (van Beek et al., 2011). A multiple linear regression model for log-transformed daily $N_2O$ fluxes gave significant effects of site and the amount of nitrate in 0 – 20 cm soil depth (both with $p < 0.001$) with highest fluxes measured at high soil nitrate. By this model, 64 % of the variation of measured $N_2O$ fluxes could be explained (Nakagawa & Schielzeth, 2013).

Soil nitrate contents are enhanced by mineral fertilizer inputs on the one hand and mineralization and nitrification of organic N in soil organic matter (SOM) or organic fertilizers on the other hand. Several $N_2O$ emission peaks at the three agricultural study sites occurred subsequent to mineral fertilizer or slurry application, especially at site AR and in spring 2012 at all three sites (Fig. 4). High soil nitrate, exceeding the current N uptake capacity of vegetation can cause increased $N_2O$ production through denitrification, thus N fertilization often leads to enhanced $N_2O$ emissions for several days to weeks (Velthof & Oenema, 1995b; Bouwman et al., 2002; Grant et al., 2006). In addition, a nitrate surplus in soil promotes incomplete denitrification and increasing $N_2O/N_2$ product ratios with the associated risk of $N_2O$ emissions (Firestone et al., 1980; Farquharson & Baldock, 2008; Senbayram et al., 2012). At site AR, strong $N_2O$ emission peaks occurred after fertilization in spring when vegetation was missing or seeded plants were emerging (Fig. 4).

Therefore, instead of relating annual $N_2O$ emissions to annual N balances, short-term N balances for about two week intervals were calculated for site AR and the vegetation periods 2012 and 2013 and related to the $N_2O$ balances of the same period. This was conducted by considering the N input by fertilizers as well as the N uptake by plants (Fig. 9). During the first weeks after fertilizer application, N surpluses of up to 99 kg $ha^{-1}$ occurred, leading to extremely high short-term $N_2O$ releases in some cases. The increasing N uptake in the subsequent periods was characterized by N balances ranging from -48 to 12 kg N $ha^{-1}$ without significant $N_2O$ emissions. These findings confirm to a meta-analysis of van Groenigen et al. (2010), who found no differences in $N_2O$ emissions for negative or slightly positive N

balances, but significantly increasing emissions for a surplus of 90 kg N $ha^{-1}$. During the period 2012/13, 73 % of $N_2O$ emissions at site AR occurred in April and May, while for the period 2013/14, 90 % of the total annual $N_2O$ budget was emitted in May. Therefore, it can be concluded that in combination with tilling, which might increase the availability of easily decomposable organic C for denitrifiers (Nykänen et al., 1995), fertilization of peat soils during periods with lacking N uptake capacity, bears the risk of substantial $N_2O$ emissions (Maljanen et al., 2003b; Regina et al., 2004).

After a second smaller fertilization peak at site AR in June 2013, $N_2O$ emissions were reduced to zero or even small uptakes of $N_2O$ were detected (Fig. 4), which can be explained by increased vegetation productivity. The growing plants act as competitor for nitrate to the denitrifiers, leading to complete denitrification as nitrate availability is strongly decreased. This was described for pristine (Roobroeck et al., 2009) or restored peatlands (Silvan et al., 2005) were N availability is usually limited (Martikainen et al., 1993). Our results suggest that on sites with very high $N_2O$ production potential, emissions can be eliminated by a continuous coverage of highly productive plants and prevention of fertilization when N uptake is limited. $N_2O$ uptake into soils is often linked to low mineral N and high moisture contents (Chapuis-Lardy et al., 2007). However, the small but continuous $N_2O$ uptakes at site AR, beginning in June 2013, were probably attributed to a high denitrification potential, stimulated by the excess of nitrate during May, and a shift to $N_2O$ consumption by denitrifiers when nitrate competition by plant roots increased (Roobroeck et al., 2009).

On average, $N_2O$-N emissions from the agricultural study sites accounted for 2.2, 5.9 and 13.2 % of applied N for sites GW, GM and AR, respectively. The values for sites GW and GM fit well with those presented by van Beek et al. (2010) for grazed grasslands on organic soil in the Netherlands with comparable GWLs. Therefore, our results support the findings of van Beek et al. (2010), who argued that mean annual GWL should be used in addition to N input for estimating $N_2O$ emissions from organic soils, as the ratio of $N_2O$ emissions to N input increases with decreasing GWLs. However, our results illustrate that the type of management should be considered as well, as arable cropping can induce a disproportional increase of $N_2O$ emissions related to N input.

Drained organic soils are known to emit significant shares of their annual $N_2O$ budget during the winter period (Priemé & Christensen, 2001; Maljanen et al., 2003b), increasing with the number of freezing and thawing cycles (Regina et al., 2004). Thereby, $N_2O$ emissions can be ~~are~~ enhanced during freezing as well as thawing, since both processes release C into the soil, which is rapidly utilized by heterotrophic denitrifiers (Koponen et al., 2006). In the present study, $N_2O$ pulses occurred during freezing events but fluxes declined rapidly after freezing. ~~In contrast, emission peaks during winter were~~ This was more pronounced when no snow cover was present as observed in the first and third winter ~~year~~ when only one period with negative temperatures occurred, but not in the second

~~year~~winter, when more freezing and thawing cycles appeared (Figs. 1 and 4). ~~The reason might be the deeper frozen soils in the first and third winters, as no snow cover was present, inducing higher C releases.~~ These results suggest that the predominating process that enhanced winter $N_2O$ fluxes was freezing rather than thawing of the peat soils. As wet peat soils have a high heat capacity, $N_2O$ fluxes did not increase directly after air temperatures became negative but few days later due to the time lag between changes in air and soil temperature. This could also explain the missing $N_2O$ pulse in the second winter as the frost could not penetrate the peat sufficiently to generate an enhanced release of C and N as a consequence of snow cover. Xu et al. (2016) demonstrated that the release of C and N during freezing as well as $N_2O$ emissions were enhanced by a lower freezing temperature, which underlines the results of this study.

Annual $N_2O$ emissions were significantly related to mean annual GWL (Fig. 10b), which might be explained by increasing amounts of nitrate in top soil with increasing drainage intensity (Fig. 11a). As the differences in soil nitrate could not be attributed to different N fertilization intensities (Table 1), the GWL seemed to control nitrification processes. Koops et al. (1997) emphasized that nitrification is an important process for $N_2O$ losses from peat soils, while Dowrick et al. (1999) stated that denitrification is the main source for $N_2O$ emissions from drained organic soils as the nitrate produced from peat mineralization is reduced in small-scale anaerobic porosity. However, both nitrification and denitrification processes likely contributed to $N_2O$ emissions as sites GM and AR showed strong fluctuations in GWL (Fig. 2), which generally leads to a pronounced cycling of both processes and thus enhanced $N_2O$ release (Goldberg et al., 2010; Jørgensen & Elberling, 2012).

### 4.3 $CO_2$ exchange and NECB

All four study sites were net C sources during the two years of $CO_2$ measurements (Table 2 and Fig. 7). Compared to IPCC (2014) emission factors for temperate organic soils, the sites showed NEE values above the given range for their respective land use categories. While the NEE of site AR was 9.0 Mg $CO_2$-C $ha^{-1}$ $a^{-1}$ in 2012/13, which is within the 95 % confidence interval of 6.5 – 9.4 Mg $CO_2$-C $ha^{-1}$ $a^{-1}$ given by IPCC (2014) for drained temperate croplands, it was above that range in 2013/14 (11.2 Mg $CO_2$-C $ha^{-1}$ $a^{-1}$). The NEE of sites GM and GW exceeded the intervals for nutrient-rich temperate grasslands that are deep-drained (5.0 – 7.3 Mg $CO_2$-C $ha^{-1}$ $a^{-1}$) or shallow-drained (1.8 – 5.4 Mg $CO_2$-C $ha^{-1}$ $a^{-1}$) in both years (Table 2). If the NECB is considered, the C losses of the agricultural sites were even higher, thus exceeding the upper values of IPCC emission factors for the respective land use categories by a factor of 2.0, 2.2 and 1.6 for sites GW, GM and AR, respectively. Moreover, the C loss from site UG clearly exceeded the average IPCC emission factor for rewetted and nutrient-rich temperate organic soils of 0.5 Mg $CO_2$-C $ha^{-1}$ $a^{-1}$ in both years.

Recently published results for utilized organic soils in the same climatic region as the study area of this observation showed net C losses of 4.3 – 8.2 Mg $CO_2$-C $ha^{-1}$ $a^{-1}$ for an intensively managed peat bog grassland in Germany (Beetz et al., 2013), 3.3 – 8.6 Mg $CO_2$-C $ha^{-1}$ $a^{-1}$ for extensively managed grasslands on *histic Gleysol* in Germany (Leiber-Sauheitl et al., 2014) and 6.9 – 16.7 Mg $CO_2$-C $ha^{-1}$ $a^{-1}$ for grassland and arable cropping on bog and fen soils in Denmark (Elsgaard et al., 2012). The highest value of 16.7 Mg $CO_2$-C $ha^{-1}$ $a^{-1}$ represented a rotational grassland on fen soil, thus a comparable system to site AR in 2013/14, which showed a similar NECB of 17.7 Mg $CO_2$-C $ha^{-1}$ $a^{-1}$. However, the NEE of the Danish site was even higher (13.6 Mg $CO_2$-C $ha^{-1}$ $a^{-1}$) than at site AR (11.2 Mg $CO_2$-C $ha^{-1}$ $a^{-1}$), indicating that C removal by harvest from site AR was comparatively high. The permanent grassland sites studied by Elsgaard et al. (2012) showed C losses between 6.9 and 10.4 Mg $CO_2$-C $ha^{-1}$ $a^{-1}$. In conclusion, C losses of sites UG, GW and AR were at the upper end of literature values, while the NECB of site GM clearly exceeded the given ranges. The comparatively high C losses of the study sites highlight the functioning of the study region as a considerable C source, underlining the need for mitigation strategies.

Seasonal variability of NEE on agricultural grasslands cannot only be explained by environmental parameters as their influence is often superposed by management activities like grassland cuttings (Wohlfahrt et al., 2008b). Land use intensity affects the NEE of ecosystems, as the frequency of biomass removals influences respiration processes as well as photosynthesis (Soussana et al., 2007). Generally, it is assumed that NEE increases with the number of cuttings, since GPP is reduced to almost zero for several days after harvest, while $R_{ECO}$ can remain high, depending on the extent of soil respiration (Schmitt et al., 2010). At the studied grassland sites, $R_{ECO}$ was often reduced by cutting events but not in the same degree as GPP, leading to sharp increases of NEE after harvest (Fig. 5). The effect of an increased number of grassland cuttings was especially pronounced at site GM, where four cuttings were conducted in the second year, compared to three cuttings in the first year. Thereby, $R_{ECO}$ was reduced to a greater extent than GPP, leading to a slightly increased NEE. However, at site GW the effect was different when the number of cuttings increased from two in the first to three in the second year. Here, a smaller $R_{ECO}$ but slightly increased GPP resulted in a lower NEE in the second year. The same effect was visible for GPP when comparing sites GM and GW for a given year (Table 2). These results suggest that changing grassland management from two to three cuttings per year did not reduce total annual photosynthetic activity, while GPP could be diminished by four cuttings. However, irrespective of total number of grassland harvests, the first cuts were performed in May, the common time for intensively managed grasslands as the average growth rate is at its maximum (Parsons & Chapman, 2000). Before the first cut, the NEE of grasslands is mainly controlled by GPP (Wohlfahrt et al., 2008a). Shifting the first cut to June or July would, therefore, increase the total productivity of first

growth period and extend the phase of net $CO_2$ uptake. However, this is hardly compatible to intensive grassland management depending on profitability (McInerney, 2000) as forage quality would be too low. After a grassland cut it took several weeks until the sites showed net $CO_2$ uptake again, often closely followed by the next cutting (Fig. 5). Therefore, the cutting regime strongly controlled the NEE of the agricultural grassland sites.

Unutilized peatland ecosystems can either be source~~s~~ or sink~~s~~ of ~~for~~ $CO_2$, depending on variables like trophic status, peat temperature, water table (Bubier et al., 1998) or vegetation composition (Leppälä et al., 2011). As the difference between uptake (GPP) and release of $CO_2$ ($R_{ECO}$) is generally small, marginal changes of these parameters can invert the NEE of a peatland between different years (Bubier et al., 1999; Griffis et al., 2000; Arneth et al., 2002). At site UG, maximum daily GPP was observed in July, followed by a decrease in August, while $R_{ECO}$ reached its maximum a few weeks later then declined to a lesser extent. This was typical as the annual course of $R_{ECO}$ is usually shifted by about one month compared to GPP (Lloyd & Taylor, 1994). Consequently, daily $CO_2$ uptake reaches its maximum in spring or early summer and a net release of $CO_2$ starts in late summer when vegetation becomes senescent and $R_{ECO}$ exceeds GPP (Bellisario et al., 1998; Parsons & Chapman, 2000). A late cutting of vegetation could delay senescence and prolong the period of plant growth at site UG, which might reduce NEE. However, Beetz et al. (2013) observed that a single cutting event shifted a rewetted and extensively used peat bog grassland from a $CO_2$ sink to a small source as annual GPP was reduced by more than annual $R_{ECO}$. This cutting was, however, conducted at the end of vegetation period and GPP did not rise again. The optimum time for a one-cutting grassland system in terms of maximizing GPP by avoiding early senescence might be in late July or early August to take advantage of both a highly productive primary growth and regrowth period. In addition, this was usually the period of lowest groundwater levels (Fig. 2), ensuring the viability of a grassland cutting as the limit for trafficability on fen soils is a GWL around -30 cm (Blankenburg et al., 2001). However, a potentially smaller NEE of a one-cut system might be offset by an increase in NECB due to biomass removal.

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

considerably increased at the end of the study period due to highly productive new established grass, a lower GWL (Fig. 2) and higher temperatures (Fig. 1).

Several studies observed increasing $CO_2$ emissions from peatland ecosystems with increasing drainage intensity (e.g. Moore & Knowles, 1989; Bubier et al., 1998; Drösler, 2005; Dinsmore et al., 2009). Since the variability of NEE for an individual agricultural site strongly depends on management (Wohlfahrt et al., 2008b) as described above, inter-site comparison is necessary to illustrate the effect of water level on NEE. On average of the four study sites and both years, NEE significantly increased by about 220 kg $CO_2$-C $ha^{-1}$ $a^{-1}$ per cm lowering of mean annual GWL (Fig. 12a). Moreover, our results suggest that arable cropping of peatlands did not lead to higher $CO_2$ emissions per se, confirming recent observations from peatland sites in Germany (Drösler et al., 2013) and Denmark (Elsgaard et al., 2012). Despite a lower mean annual GWL on site AR (Table 1), NEE and NECB of sites AR and GM did not differ significantly (Fig. 7). This can be explained by a lower $R_{ECO}$ due to missing vegetation cover and water logging after harvest at site AR in the first year and a very high GPP due to undersown grass in the second year. Furthermore, Estop-Aragonés et al. (2012) argue that in compacted peat soils with high bulk densities and ash contents, oxygen penetration is reduced compared to less compacted soils, resulting in lower air filled porosity and soil respiration. Due to the higher peat degradation of site AR (Table 1), this could partly explain the similar NEE of sites AR and GM.

While Aurela et al. (2007) reported that a drought period in a Finnish sedge fen increased $R_{ECO}$ and thus NEE, Leppälä et al. (2011) concluded that the difference in NEE between wet and dry years for natural peatlands in Finland resulted from alterations of GPP rather than $R_{ECO}$. For the dryer second year of our observations, $R_{ECO}$ of site UG was lower than in the first year, while GPP decreased only marginally (Table 2). However, comparing only July and August, the period with greatest difference in GWLs 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 6.8 Mg $CO_2$-C $ha^{-1}$ in 2012 and 2013, respectively), which is in line with results presented by Parmentier et al. (2009) and Muhr et al. (2011). GPP slightly increased in the drier year (-7.2 and -7.6 Mg $CO_2$-C $ha^{-1}$ in 2012 and 2013, respectively). As main reason for differences in NEE between the two years, the weather conditions in spring could be identified. In 2012, the growing season, calculated by the method of Janssens (2010), started on 20 March, while it was delayed by more than one month in 2013 to 23 April. As a result of different weather conditions, cumulated $R_{ECO}$ for April and May was 4.1 Mg in 2012 and only 2.2 Mg $CO_2$-C $ha^{-1}$ in 2013. Besides, GPP was -3.2 Mg for April and May 2012 and -2.3 Mg $CO_2$-C $ha^{-1}$ in 2013. These differences cannot be explained by mean GWL for the two months (-4.9 cm in 2012 and -8.4 cm in 2013) as the different weather conditions were the dominating parameter. Thus, respiration processes were stimulated more than plant productivity by the earlier start of growing season, indicating that shorter winter periods potentially increase the risk of higher C losses

from peatland ecosystems. Griffis et al. (2000) studied the NEE of a subarctic fen and concluded that the phenological stage of vegetation relative to the climatic conditions is important for interannual variability of NEE. In conclusion, the mean GWL of single years cannot be solely used to predict the variability of NEE at the same site or between sites with different management as climatic and management effects can be of dominating importance.

### 4.4 Global warming potential

The global warming potential (GWP) of the four study sites increased in the same order as NEE and NECB. However, the difference between sites GW and AR was significant for GWP whereas it was not significant for NEE and NECB (Fig. 7). This can be explained by significantly higher $N_2O$ emissions at site AR (Fig. 6b). NEE mainly controlled the GHG balances, accounting for 72, 69, 66 and 59 % of the GWP on sites UG, GW, GM and AR, respectively. In addition, the balances of C export via harvest and C import via slurry contributed considerably to the GWP of the agricultural sites, accounting for 21, 23 and 27 % for sites GW, GM and AR, respectively, indicating a higher share of anthropogenic C fluxes with higher land use intensity. Compared to other observations or reviews of peatland GHG emissions in northern or temperate Europe, the GWP of the study sites was at the upper end of presented emission factors (Nykänen et al., 1995; Langeveld et al., 1997; Kasimir-Klemedtsson et al., 1997; Alm et al., 2007a; Oleszczuk et al., 2008; Maljanen et al., 2010; Drösler et al., 2013).

Site UG showed a significantly lower GWP compared to the agricultural sites, supporting the assumption that rewetting of drained organic soils reduces their climatic footprint (Höper et al., 2008; Beetz et al., 2013). The lower GWP of site UG was a result of missing C losses through harvest and reduced $CO_2$ and $N_2O$ emissions that could mainly be attributed to the high GWLs (Fig. 12b), outweighing the higher $CH_4$ release (Fig. 6a). A linear regression for all four sites and both years gave a significant increase of GWP for about 410 kg $CO_2$-C-eq $ha^{-1}$ $a^{-1}$ per cm lowering of mean annual GWL (Fig. 12b). The higher slope compared to NEE (Fig. 12a) was a result of $N_2O$ emissions, significantly increasing with drainage intensity as well (Fig. 10b). However, as $CH_4$ emissions tended to increase exponentially when water levels were close to the soil surface (Fig. 10a), the slope might decline or even invert for a mean annual GWL around or above 0 (Augustin & Joosten, 2007). Therefore, the intercept of ~2 Mg $CO_2$-C-eq $ha^{-1}$ $a^{-1}$ should not be over-interpreted. A mean annual GWL of about 10 cm below the soil surface is often referred to as an optimum scenario for mitigating GHG emissions from peatlands, as $CO_2$ emissions are greatly reduced or even negative (i.e. $CO_2$ uptake) and $CH_4$ fluxes are hampered by the small oxic horizon (e.g. Couwenberg et al., 2011). However, this is not only controlled by mean annual GWL, but equally by groundwater fluctuations (Dinsmore et al., 2009). Thus, the relatively high GWP of site UG (3.8 Mg $CO_2$-C-eq $ha^{-1}$ $a^{-1}$) in spite of a high mean annual

GWL (Table 1) suggests that a further increase and stabilization of water levels might be necessary to reduce the climatic impact of that site. The lack of natural, peat forming mire vegetation (Table 1) supports this assumption as the GWP of natural or rewetted reed and sedge fens is assumed to be around 1 Mg $CO_2$-C-eq $ha^{-1}$ $a^{-1}$ (Couwenberg et al., 2011; Drösler et al., 2013).

### 4.5 Yield related emissions

To assess the climatic footprint of the agricultural study sites, their function in terms of forage and milk production has to be considered in addition to area related GHG emissions. On average of two years, site GW represented the most climate-efficient forage production system of the three sites, whereas site AR caused the greatest GHG emissions relative to energy yield (Table 4). Observations of greenhouse gas emissions from arable forage cropping systems at two sites on mineral soil in northern Germany resulted in yield related emissions between -18 and 32.5 kg $CO_2$-C-eq $(GJ\ NEL)^{-1}$, including all emissions during crop production, transport and storage (Herrmann et al., 2014). Hence, the field based emissions at the study sites presented here, demonstrate that forage produced on fen soils is burdened with many times higher GHG emissions compared to forage from mineral soils of the same region.

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

Comparing the two grassland sites, the four-cut system of site GM in 2013 showed the highest and the three-cut system of site GW in 2013 induced lowest yield related GHG emissions. In addition, the two-cut system of site GW in 2012 had higher yield related emissions than the three-cut systems of both sites. Therefore, a three-cut grassland in combination with a preferably high GWL represented the most climate-efficient management system at the studied fen soils. On average of both years, the energy yield of site GW was 19 % lower compared to site GM, while the GWP was 34 % lower. This difference was only significant for GWP. Thus, the effect of a raised water level can be assumed to be greater for GHG emission reduction than for yield reduction. This is in line with results of Renger et al. (2002), who

reported that for a mean GWL of -30 cm, 90 % of optimum plant output can be reached, while GHG emissions can be reduced for 40 – 50 % of maximum emissions. These values were obtained by a water regime model and represent an optimum scenario, indicating that further potential exists to improve the climate efficiency of forage production on site GW. Reasons for reduced productivity on poorly drained soils could include the loss of sown species in favor of undesirable species with increasing sward age (Hopkins & Green, 1979) and a lower soil warming in spring due to high soil moisture, resulting in delayed plant growth (Tyson et al., 1992). The first aspect was evident in increasing shares of creeping bentgrass (*Agrostis stolonifera*) and water foxtail (*Alopecurus geniculatus*) at site GW, indicating the need for occasional resowing of productive species like perennial ryegrass (*Lolium perenne*).

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

## 5 Conclusions

Long-term drainage intensity was the most important controlling factor for GHG emissions from the studied fen soils. NEE dominated the GHG balances of all sites and as assumed, considerable differences in GHG fluxes and balances were observed among the sites. After 20 years of rewetting (site UG), emissions of $CO_2$ and $N_2O$ were significantly lower while significantly higher amounts of $CH_4$ were emitted compared to the agricultural sites. Also, the GWP of site UG was significantly reduced. However, the site still acted as a C source and showed substantial $N_2O$ emissions, indicating that rewetting had not yet restored the natural peatland functioning as a sink for C and a negligible source for $N_2O$. Restoration progress could be promoted by a year-round stabilization of GWL close to the soil surface. In the current state, a mulching of vegetation in summer might increase total annual productivity by avoiding early senescence and thus reduce C losses.

Arable forage production (site AR) did not induce higher C losses compared to intensive grassland management and only showed a significantly higher GWP than the wet grassland site (GW) as the influence of drainage intensity was of dominating importance. However, interannual on-site variability was additionally affected by management and climatic factors. The beginning of growing season was identified as a critical period, with higher $CO_2$ losses occurring with an early start of vegetation period.

Yield related GHG emissions increased with increasing drainage and land use intensity in the order GW, GM and AR, with a significant difference between sites GW and AR.

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

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Table 1. Soil and land use characterization of the experimental sites (UG: unutilized grassland, GW: grassland 'wet', GM: 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 |
| $C_{org}$ (%)[a] | 35.0 (2.6) | 37.4 (3.9) | 17.9 (2.9) | 13.3 (1.9) |
| C/N[a] | 17.7 (1.0) | 15.7 (0.6) | 12.4 (0.4) | 12.2 (0.2) |
| Ash (%)[a] | 36.8 (11.7) | 33.6 (6.3) | 68.7 (2.3) | 74.0 (4.2) |
| Bulk density ($g\ cm^{-3}$)[a] | 0.20 (0.05) | 0.32 (0.07) | 0.54 (0.08) | 0.67 (0.09) |
| C stock ($Mg\ ha^{-1}$)[a] | 215 (57) | 361 (82) | 289 (45) | 266 (38) |
| $N_{min}$ ($kg\ ha^{-1}$)[b] | 20.8 (8.8) | 44.7 (22.7) | 73.1 (37.3) | 65.3 (31.4) |
| $NO_3$-N/$NH_4$-N[b] | 0.10 (0.16) | 0.25 (0.27) | 0.67 (0.61) | 2.55 (3.34) |
| Soil moisture ($kg\ kg^{-1}$)[b] | 2.84 (0.44) | 2.36 (0.61) | 1.15 (0.30) | 0.79 (0.16) |
| pH ($CaCl_2$)[c] | 4.58 (0.13) | 4.41 (0.18) | 5.06 (0.13) | 5.31 (0.29) |
| Groundwater level (cm)[d] | -10.9 (3.5) | -21.4 (4.6) | -33.0 (9.4) | -39.4 (4.2) |
| Fertilization ($kg\ N\ ha^{-1}\ a^{-1}$)[e] | — | 300 (240 – 400) | 260 (230 – 320) | 150 (130 – 170) |
| Type of fertilizer[f] | — | cattle slurry, CAN, ASN | cattle slurry, CAN | cattle slurry, DAP, CAN |
| Dominant plant species | Purple small-reed (*Calamagrostis canescens*), Reed canary grass (*Phalaris arundinacea*), Common rush (*Juncus effusus*) | Perennial ryegrass (*Lolium perenne*), Rough bluegrass (*Poa trivialis*), Creeping bentgrass (*Agrostis stolonifera*) | Italian ryegrass (*Lolium multiflorum*), Perennial ryegrass (*Lolium perenne*), Rough bluegrass (*Poa trivialis*) | Maize (*Zea mays*), Barley (*Hordeum vulgare*), Wheat (*Triticum aestivum*), Perennial ryegrass (*Lolium perenne*) |

[a] Given values are for 0 – 30 cm soil depth. $C_{org}$ and C/N: mean value from biannual samplings during the period May 2011 – March 2014 (n = 7). Bulk density and C stock: mean value of soil samples taken in May 2013 (n = 4). Ash content: mean value from samples taken in October 2013 (n = 4). [b] Mean value of mineral nitrogen ($NO_3^-$ and $NH_4^+$) and gravimetric soil moisture content in 0 – 20 cm soil depth from biweekly samplings during the period April 2011 – April 2014 (n = 73). [c] Mean value of two samplings in the beginning (May 2011) and in the end (July 2014) of the study (n = 8). [d] Mean value of linear interpolated weekly measurements in the period April 2011 – March 2014 (n = 4). [e] Sum of applied nitrogen from organic and mineral fertilizers on average of 2011, 2012 and 2013 and the range between the years. [f] CAN = calcium ammonium nitrate, ASN = ammonium sulphate nitrate, DAP = diammonium phosphate.

Table 2. Annual budgets of $CO_2$ exchange ($R_{ECO}$, GPP and NEE), $CH_4$ and $N_2O$ fluxes, net ecosystem carbon balance (NECB) and global warming potential (GWP) for 100 years (IPCC, 2007) for different study periods (each period is April – March). NECB is calculated from NEE and $CH_4$-C as well as slurry-C and harvested C. The GWP includes $CO_2$-C-equivalents of NEE, $CH_4$-C, $N_2O$-N, slurry-C and harvested C. Small deviations in NEE are caused by rounding. Values are annual sums and standard errors (in brackets).

| Site | Period | $R_{ECO}$ | GPP | NEE | $CH_4$ | | $N_2O$ | | NECB | GWP |
|---|---|---|---|---|---|---|---|---|---|---|
| | | (Mg $CO_2$-C $ha^{-1}$) | | | (kg C $ha^{-1}$) | (Mg $CO_2$-C-eq $ha^{-1}$) | (kg N $ha^{-1}$) | (Mg $CO_2$-C-eq $ha^{-1}$) | (Mg C $ha^{-1}$) | (Mg $CO_2$-C-eq $ha^{-1}$) |
| UG | 2011/12 | — | — | — | 38.6 (10.6) | 0.35 (0.10) | 3.3 (1.2) | 0.43 (0.15) | — | — |
| | 2012/13 | 20.7 (2.3) | -16.8 (3.1) | 3.9 (3.8) | 99.5 (47.8) | 0.91 (0.44) | 2.3 (0.4) | 0.29 (0.05) | 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) | 0.25 (0.09) | 4.5 (1.1) | 0.57 (0.13) | 1.7 (1.2) | 2.5 (1.3) |
| GW | 2011/12 | — | — | — | 6.6 (1.7) | 0.06 (0.02) | 4.7 (1.1) | 0.59 (0.14) | — | — |
| | 2012/13 | 26.6 (0.2) | -17.5 (0.3) | 9.1 (0.4) | 16.5 (2.8) | 0.15 (0.03) | 8.9 (2.0) | 1.14 (0.26) | 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) | 0.16 (0.11) | 5.9 (1.0) | 0.76 (0.13) | 8.3 (1.1) | 9.2 (1.3) |
| GM | 2011/12 | — | — | — | 1.1 (0.6) | 0.01 (0.01) | 11.4 (3.0) | 1.45 (0.38) | — | — |
| | 2012/13 | 29.6 (1.1) | -18.3 (0.4) | 11.4 (1.1) | 1.5 (0.8) | 0.01 (0.01) | 12.5 (1.9) | 1.59 (0.24) | 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) | < 0.01 (0.01) | 19.3 (4.7) | 2.46 (0.60) | 15.7 (1.5) | 18.2 (2.9) |
| AR | 2011/12 | — | — | — | 0.8 (1.8) | 0.01 (0.02) | 20.0 (1.8) | 2.55 (0.26) | — | — |
| | 2012/13 | 24.7 (1.3) | -15.7 (0.9) | 9.0 (1.5) | 4.5 (0.7) | 0.04 (0.01) | 15.3 (1.4) | 1.95 (0.17) | 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) | < 0.01 (< 0.01) | 21.6 (2.1) | 2.76 (0.27) | 17.7 (3.0) | 20.5 (3.3) |

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

| Site | Year | DM ($Mg\ ha^{-1}\ a^{-1}$) | | C ($Mg\ ha^{-1}\ a^{-1}$) | | N ($kg\ ha^{-1}\ a^{-1}$) | | NEL ($GJ\ ha^{-1}\ a^{-1}$) | |
|---|---|---|---|---|---|---|---|---|---|
| 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 |

Table 4. Annual GHG balances ($CO_2$, $CH_4$ and $N_2O$ fluxes, slurry-C and harvested C) of the three agricultural study sites in relation to energy yield (net energy lactation, NEL). Different capital letters indicate significant differences between the sites ($p < 0.05$). Values in brackets are standard errors ($n = 3$).

| Period | Site | | | | | |
|---|---|---|---|---|---|---|
| | GW | | GM | | AR | |
| | kg $CO_2$-C-eq (GJ NEL)$^{-1}$ | | | | | |
| 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 |

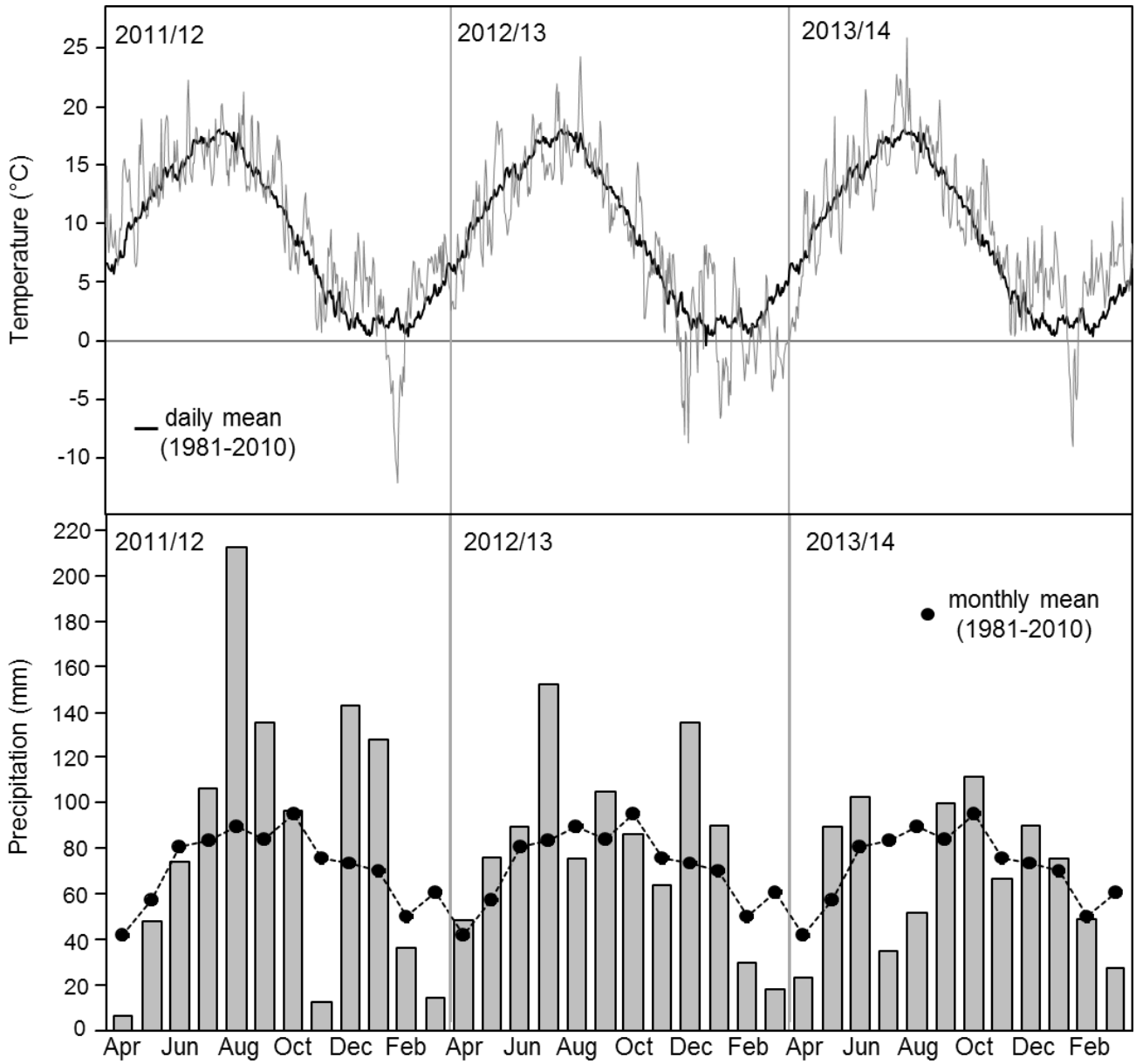


Figure 1. Daily mean air temperatures (grey line) and monthly mean precipitation sums (grey bars) during the study period (April 2011 – March 2014) compared to the long-term averages.

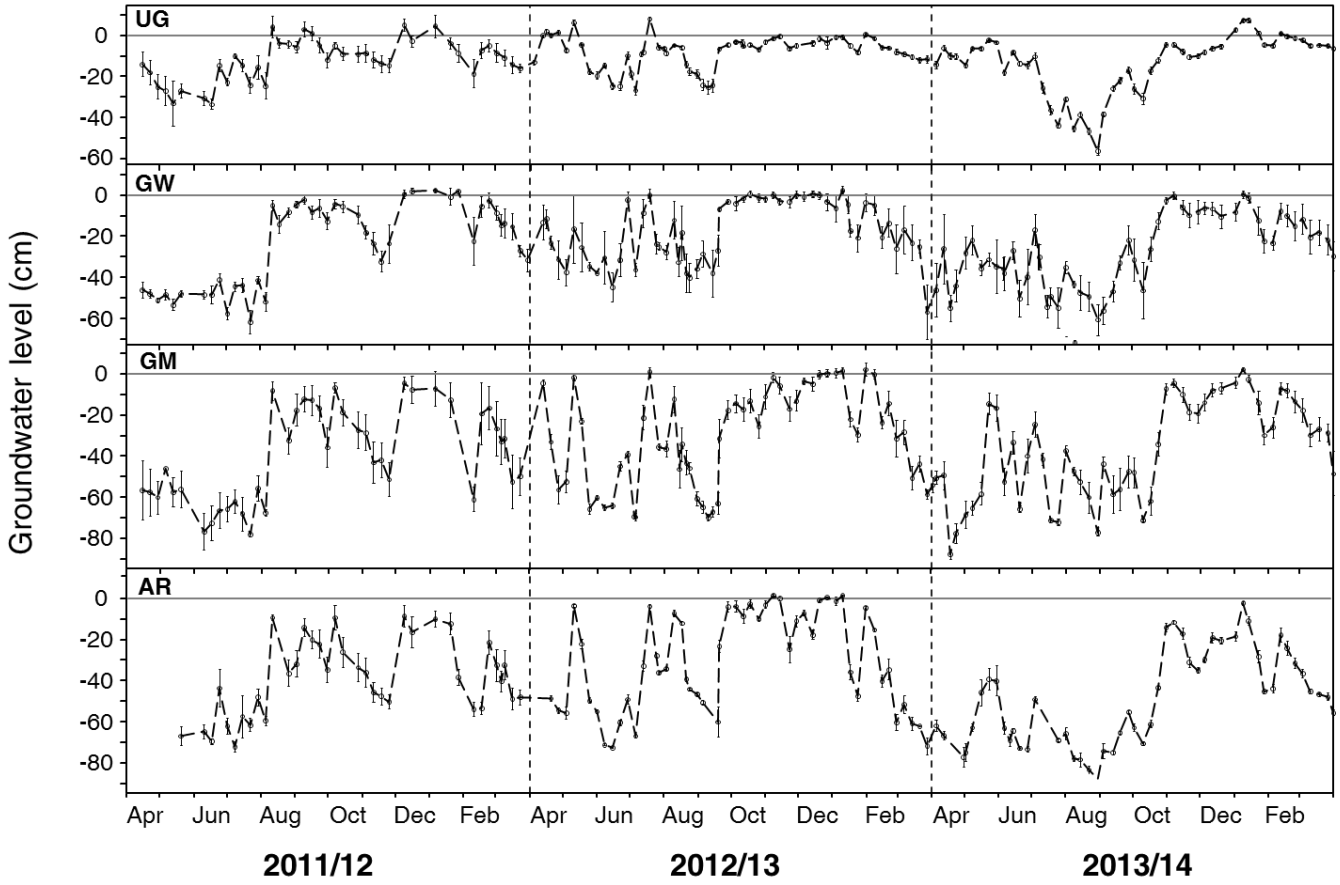


Figure 2. Development of groundwater levels (GWLs) at the four study sites during the study period (April 2011 – March 2014). Displayed are mean values ± standard errors of the manually recorded GWLs during gas flux measurements (n = 4).

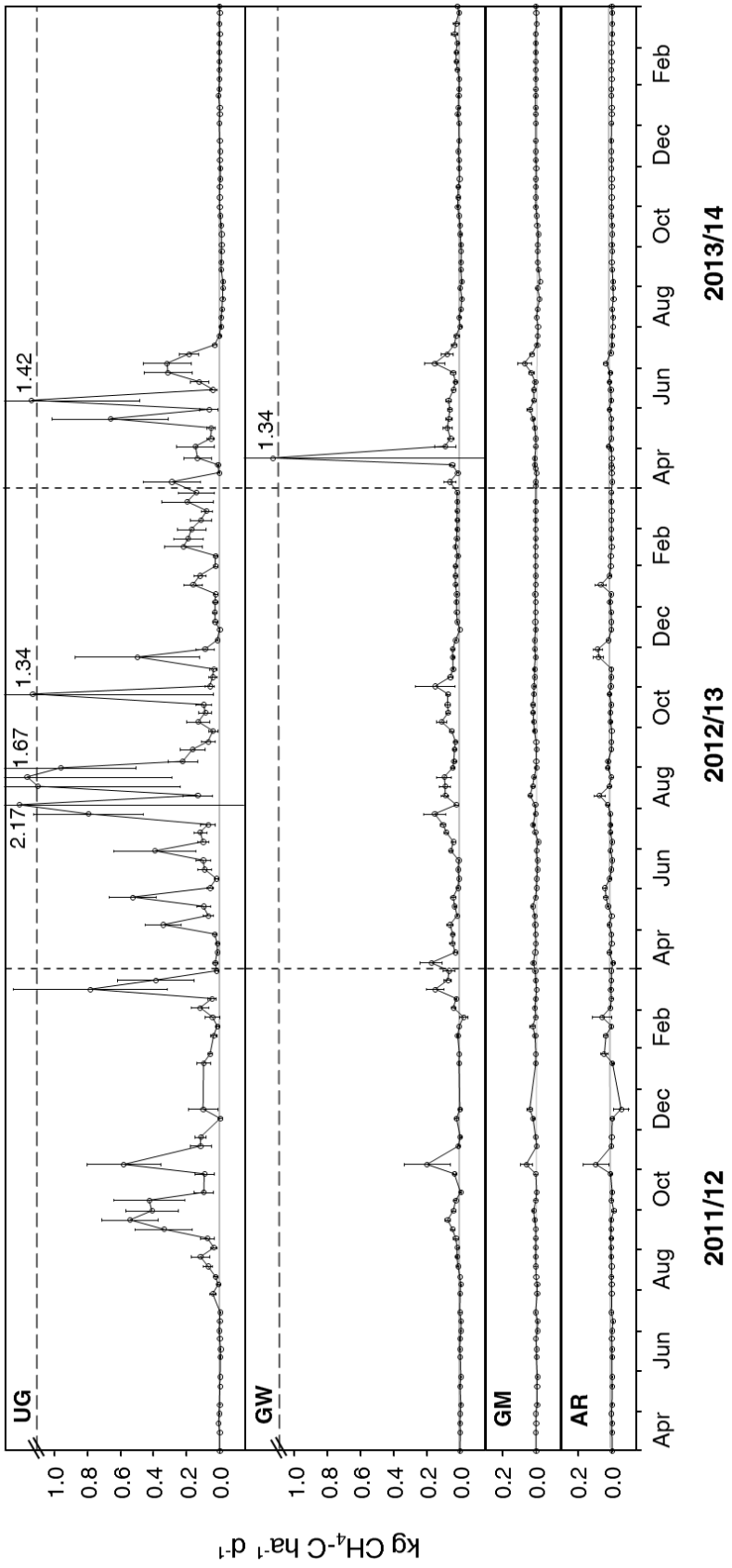


Figure 3. $CH_4$ exchange at the four study sites during the study period (April 2011 – March 2014). Values are displayed as mean ± standard error (n = 8). Note the broken y-axis for sites UG and GW.

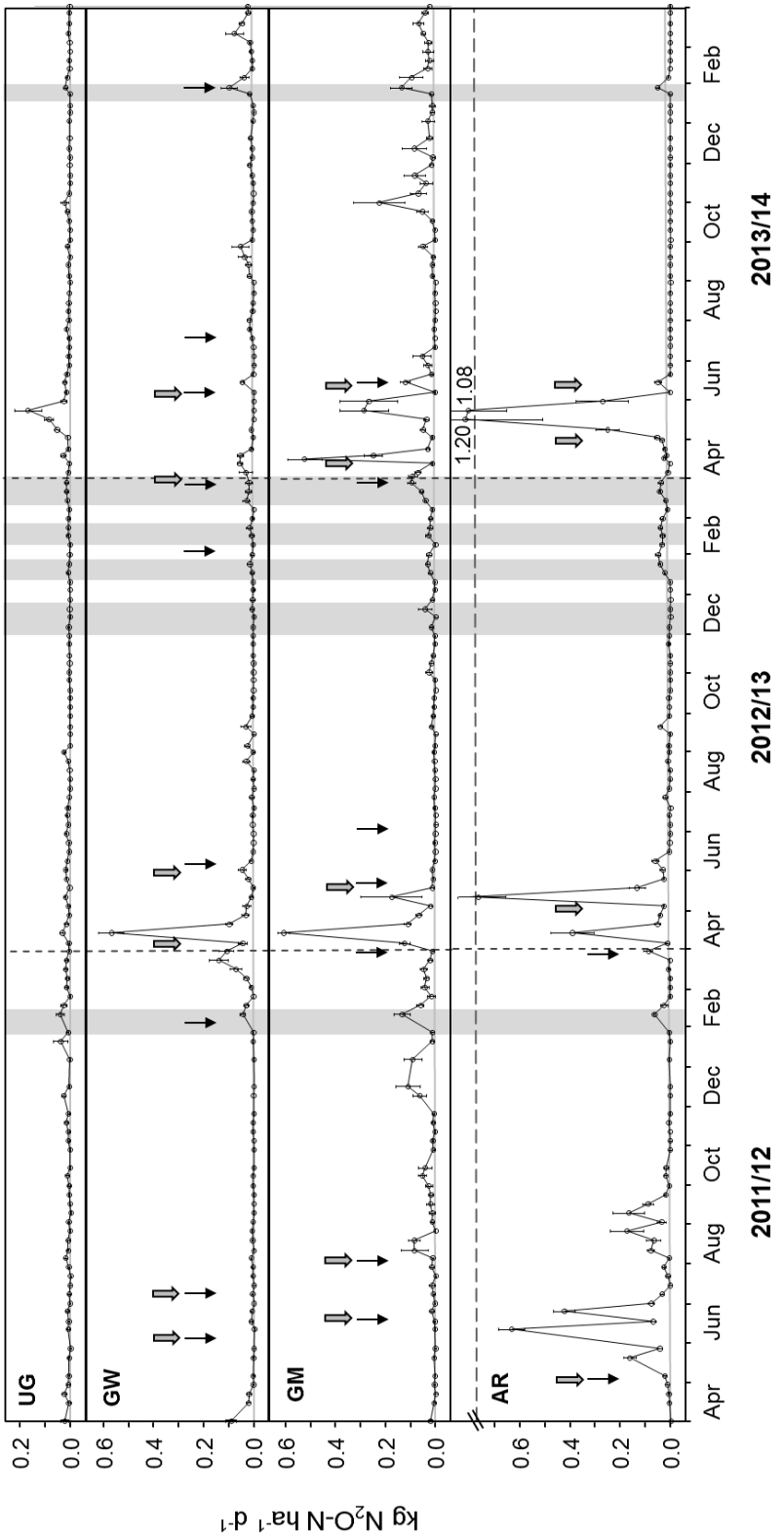


Figure 4. $N_2O$ exchange at the four study sites during the study period (April 2011 – March 2014). Values are displayed as mean ± standard error (n = 8). Note the broken y-axis for site AR. Arrows indicate applications of slurry (black) and mineral nitrogen fertilizer (grey). Grey background represents periods with mean daily temperatures below 0 °C.

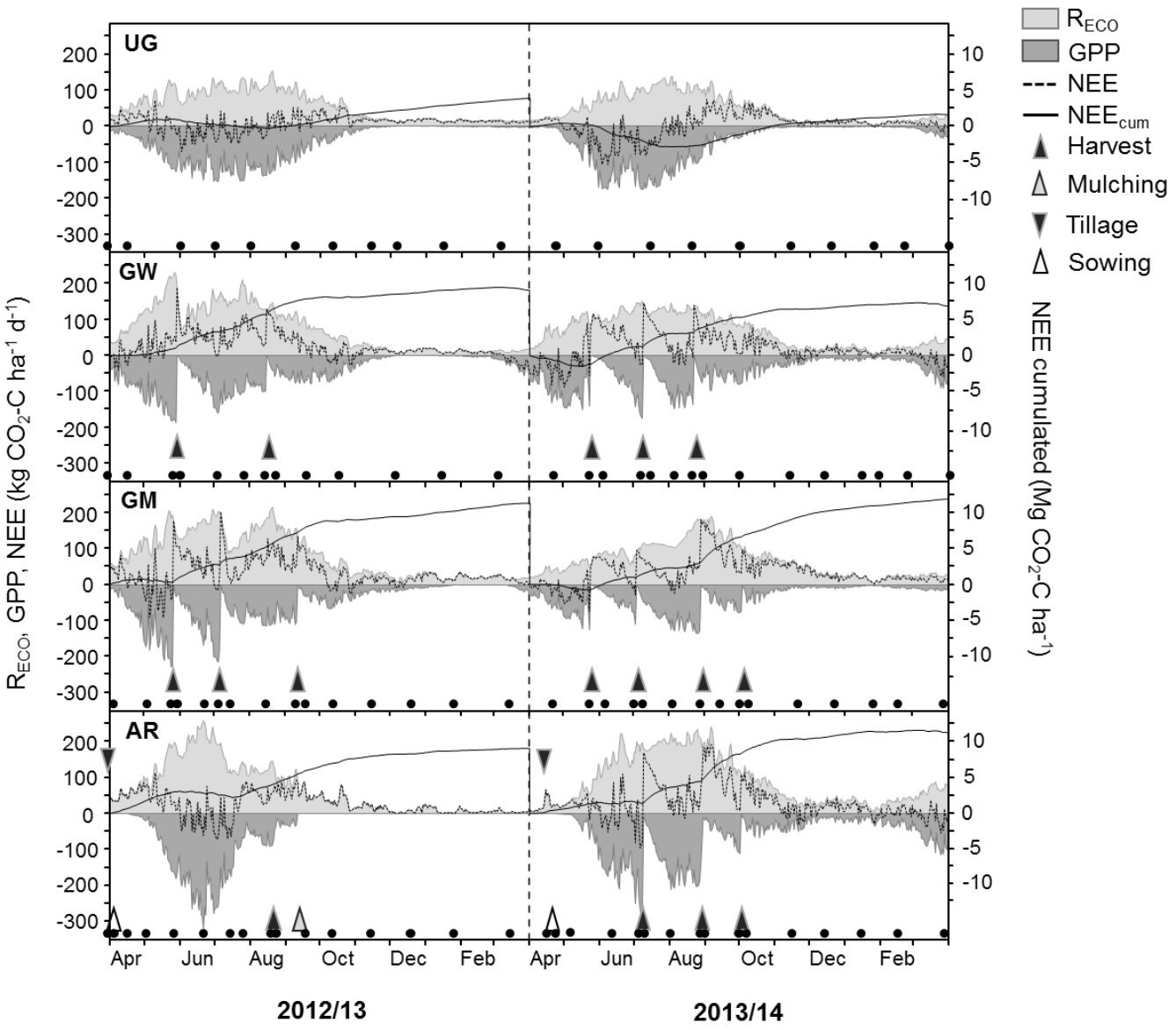


Figure 5. $CO_2$ exchange at the four study sites during a two-year study period (April 2012 – March 2014). Values are displayed as daily means of the model output (n = 3). The black continuous line shows the cumulated NEE for one year. The black dots represent $CO_2$ measurement campaigns.

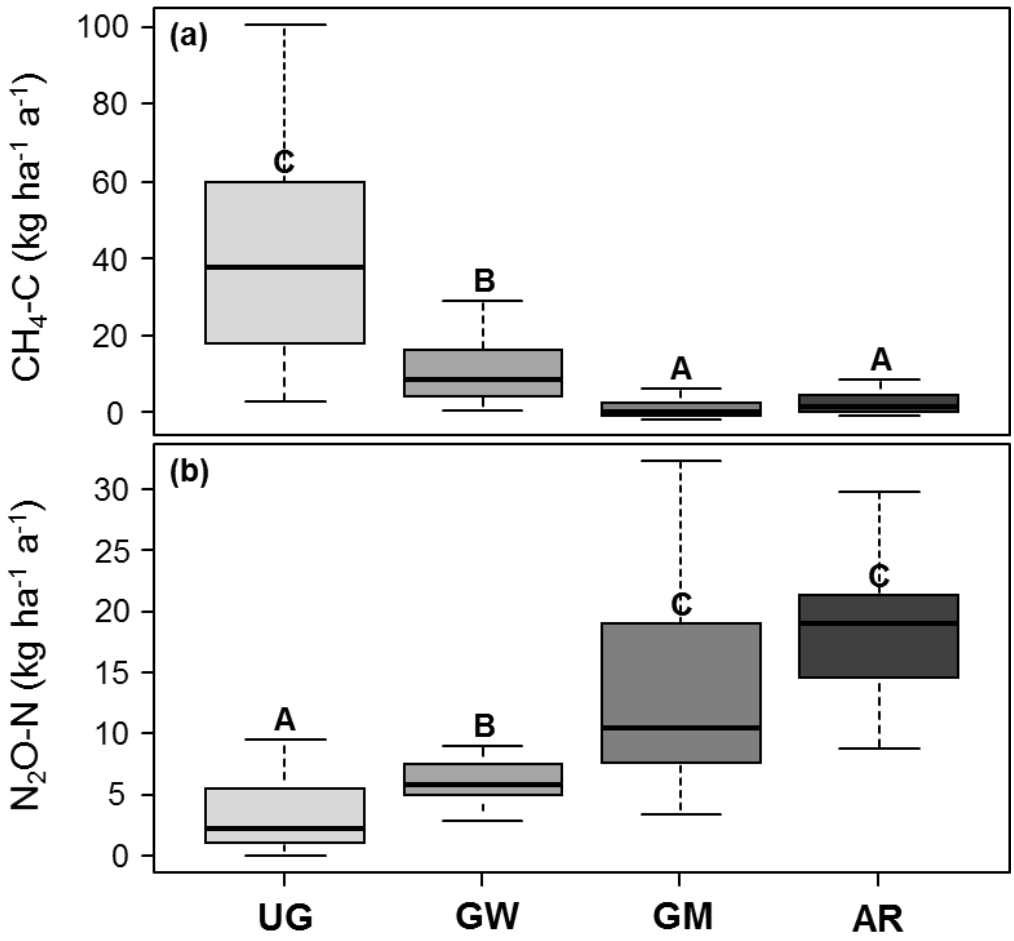


Figure 6. Annual emissions of $CH_4$ (a) and $N_2O$ (b) at the four study sites combined for three years (April 2011 – March 2014) of measurement (n = 24). Different capital letters indicate significant differences between the sites ($p < 0.05$).

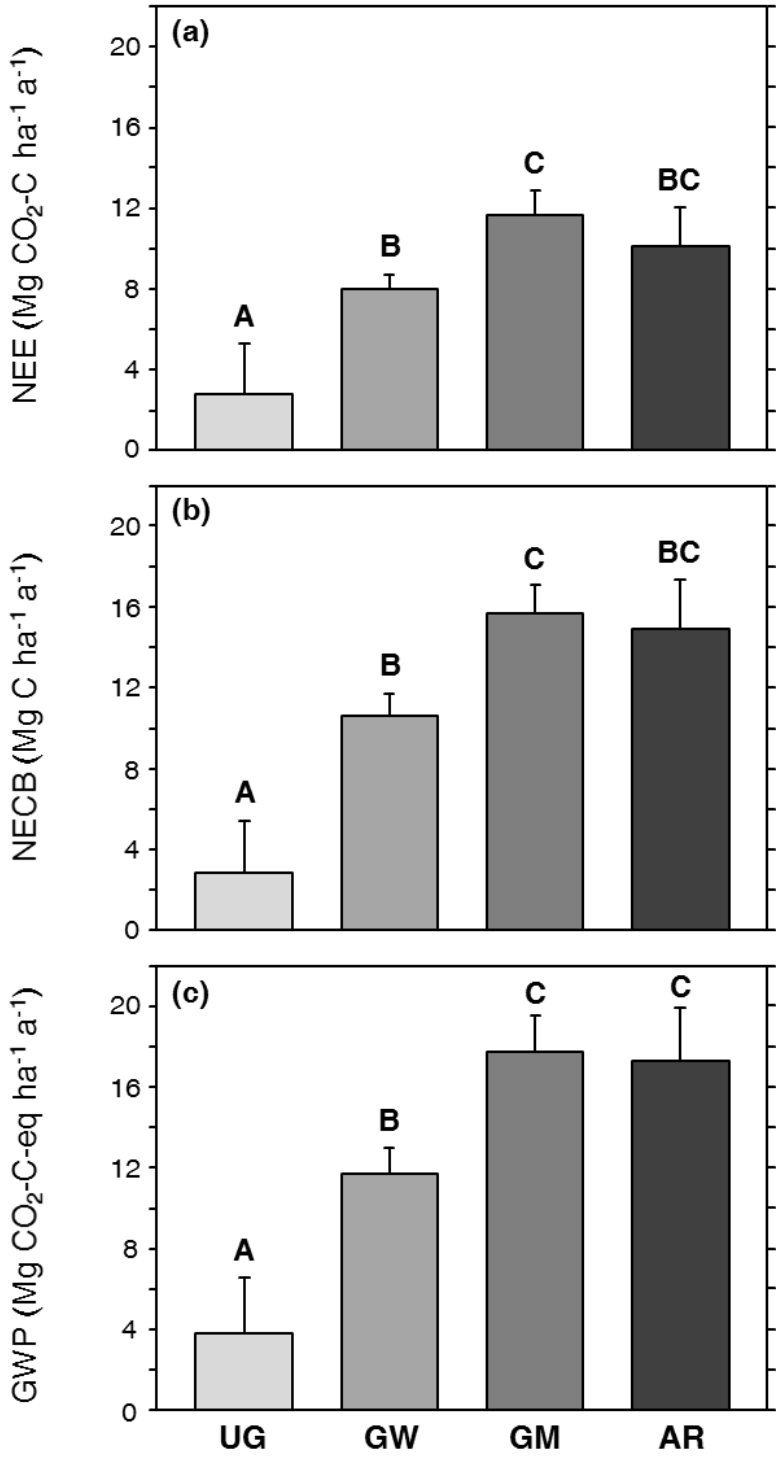


Figure 7. Mean annual budgets of net ecosystem exchange (NEE) of $CO_2$ (a), net ecosystem carbon balance (NECB) (b), and global warming potential (GWP) (c) at the four study sites for the period April 2012 – March 2014 (n = 6). Error bars represent standard error. Different capital letters indicate significant differences between the sites ($p < 0.05$).

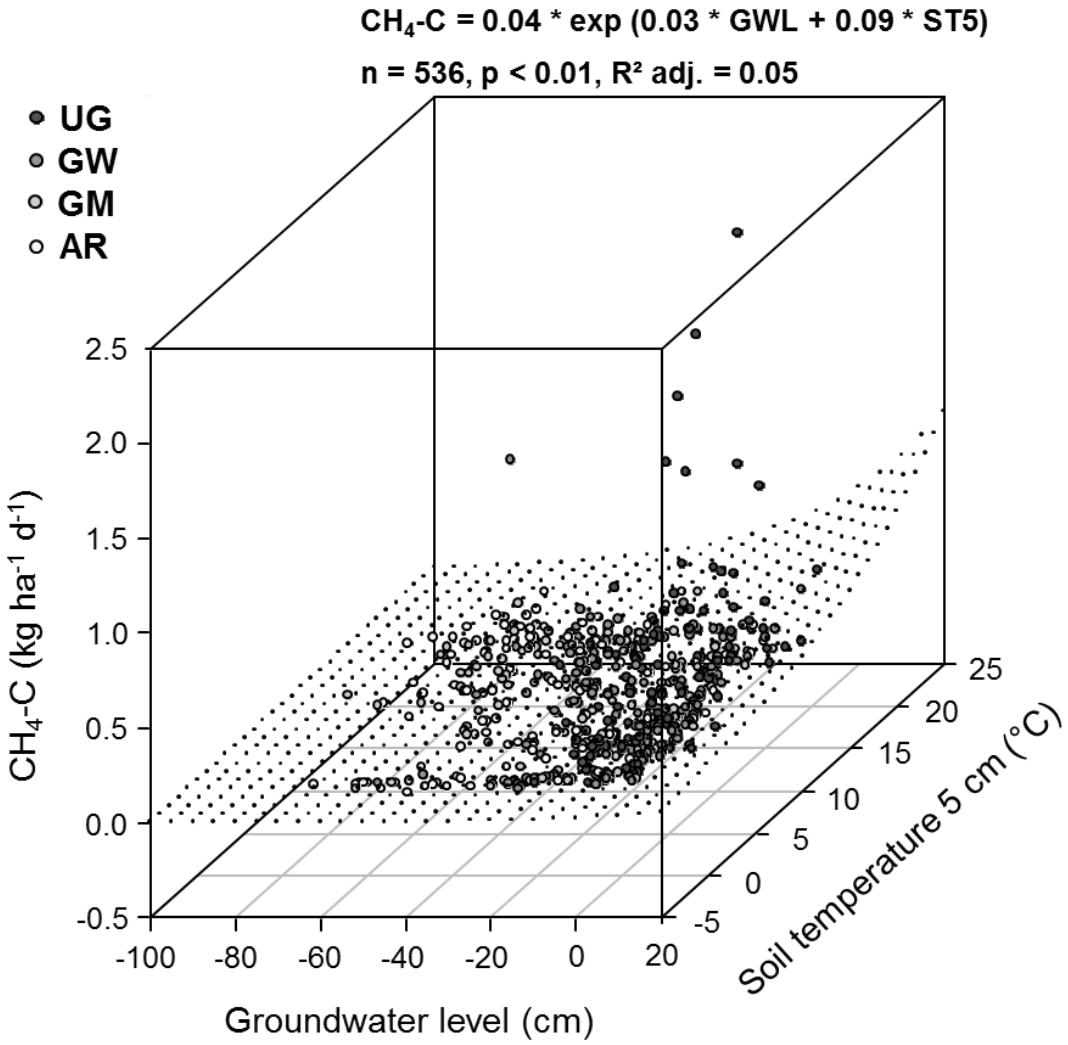


Figure 8. Relationship of daily $CH_4$ fluxes to groundwater level and mean daily soil temperature at 5 cm depth. GWL in the equation is groundwater level (cm) and ST5 is soil temperature at 5 cm depth (°C). $R^2$ adjusted was estimated for predicted versus obtained values.

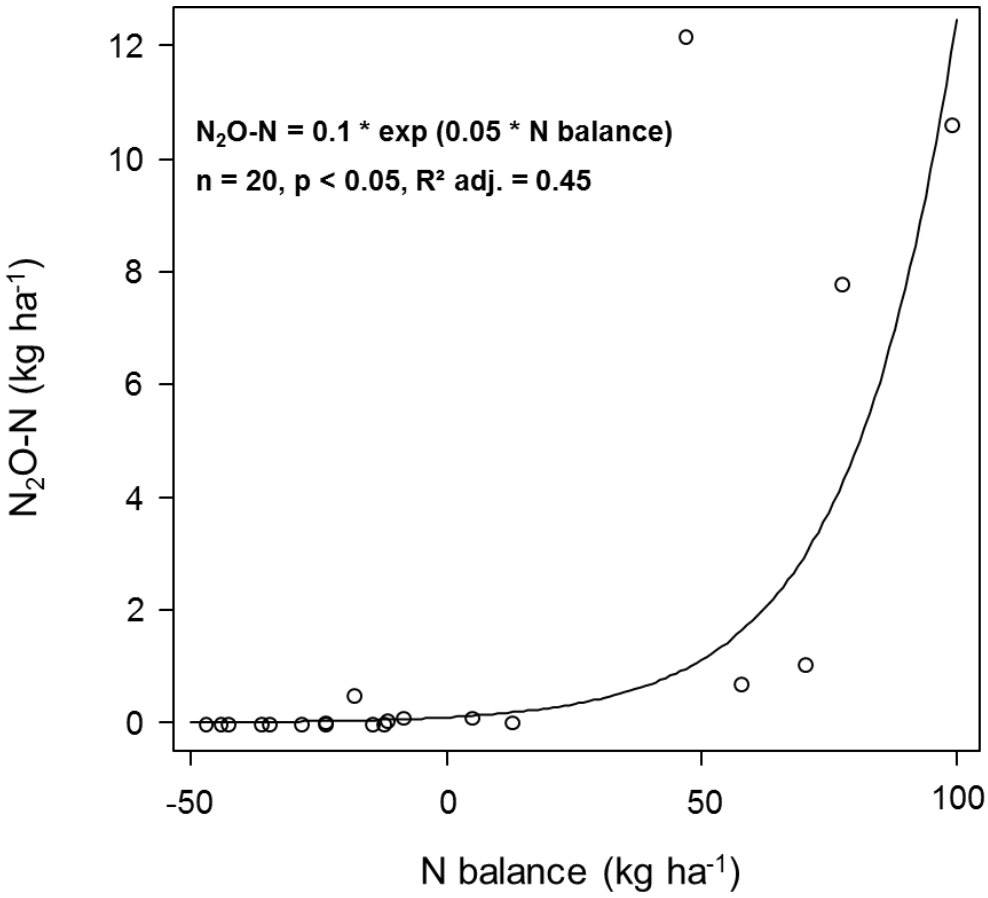


Figure 9. Relationship of cumulated $N_2O$ fluxes (n = 8) for a certain period of the growing seasons 2012 and 2013 at the arable site (AR) to nitrogen balance (n = 3) for the same period, calculated from mineral N input of mineral and organic fertilizers and the N removal by plants. $R^2$ adjusted was estimated for predicted versus obtained values.

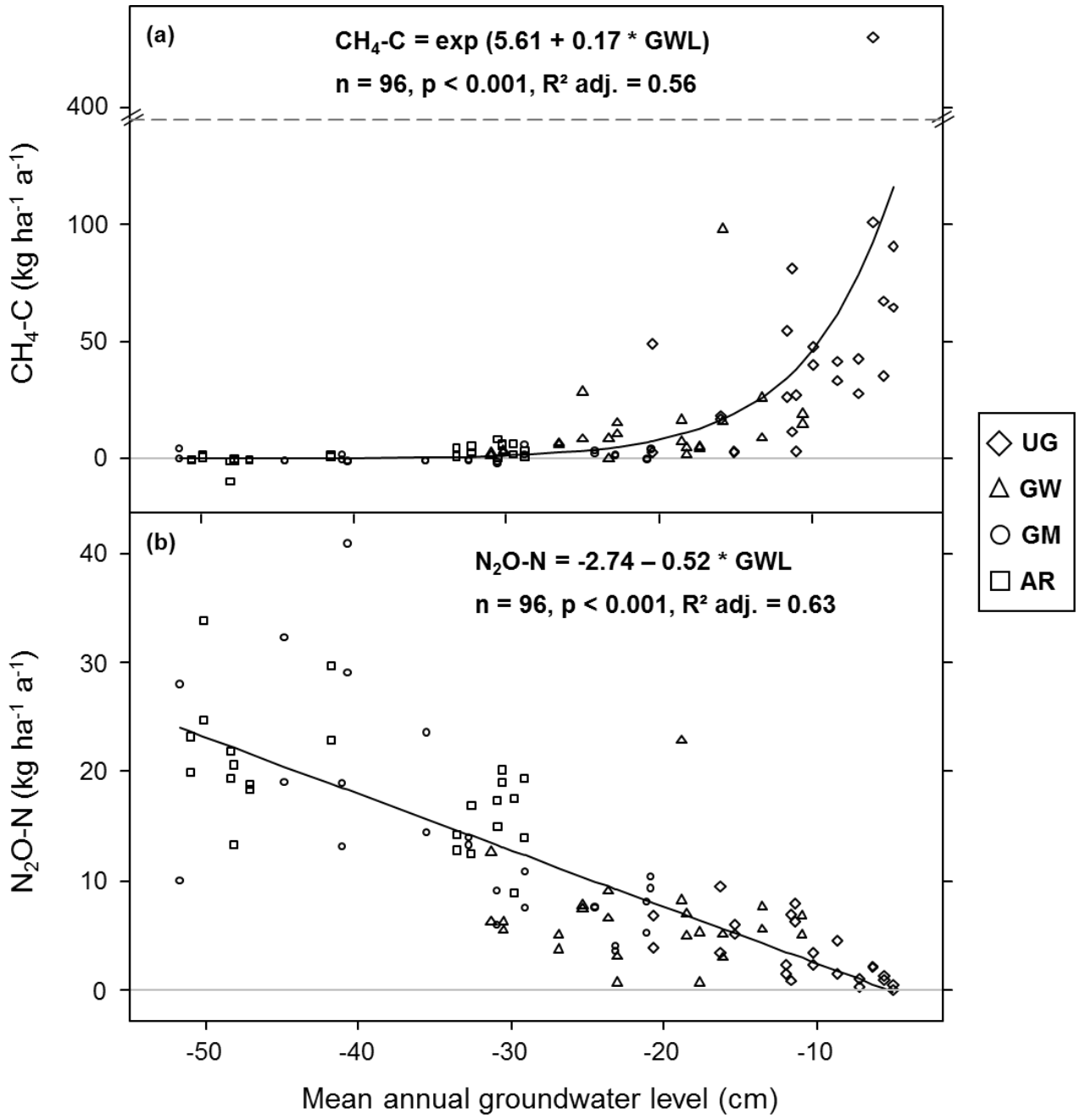


Figure 10. Relationships of cumulated annual $CH_4$ (a) and $N_2O$ fluxes (b) to mean annual groundwater level for the study period (April 2011 – March 2014) with n = 8 per site and year. GWL in the equations is mean annual groundwater level (cm). $R^2$ adjusted for exponential regression (a) was estimated for predicted versus obtained values. Note the broken y-axis for figure (a).

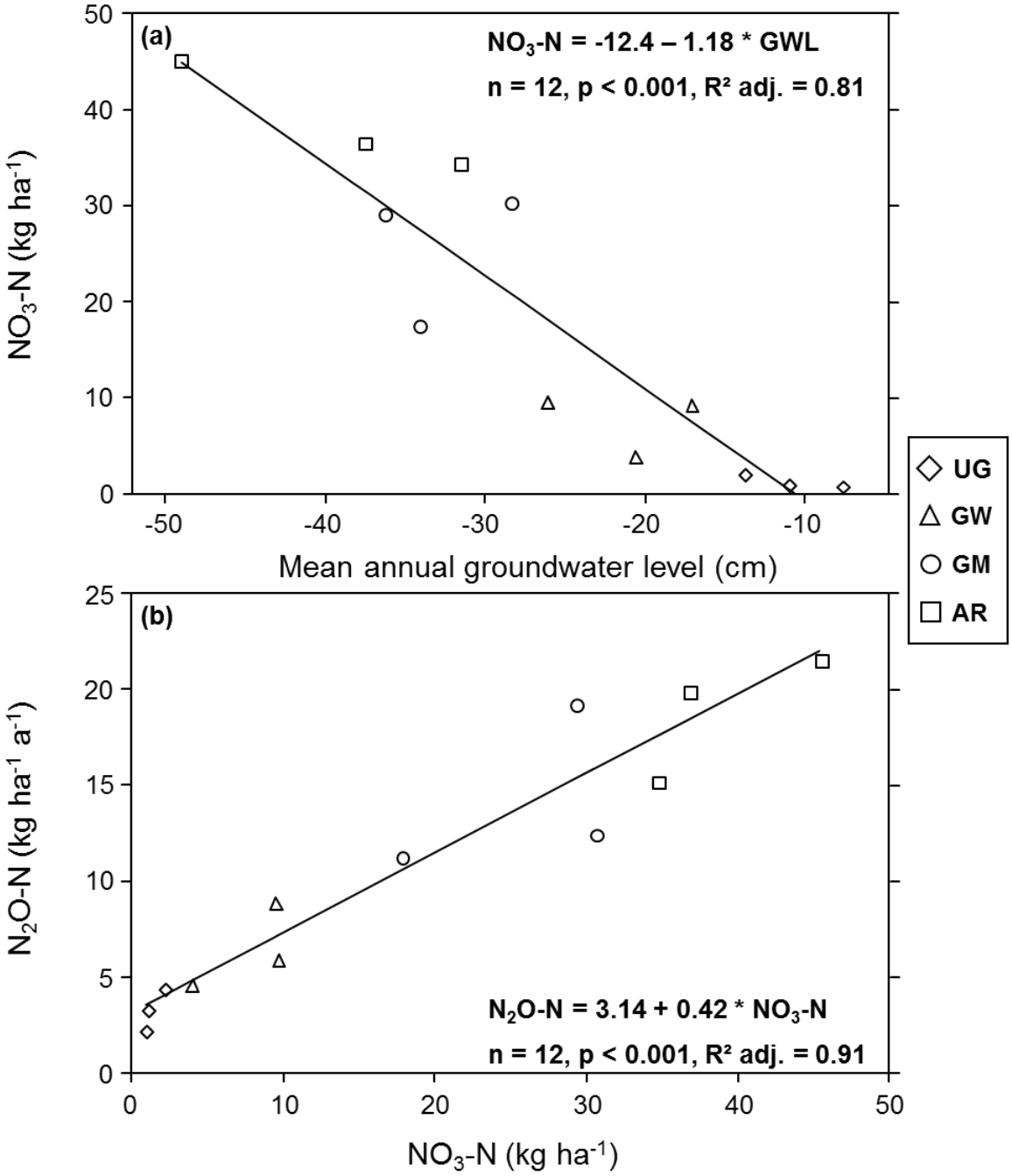


Figure 11. Relationship of mean annual amount of nitrate in 0 – 20 cm soil depth to mean annual groundwater level (a) and relationship of mean annual $N_2O$ balances to mean annual amount of nitrate in 0 – 20 cm soil depth (b). GWL in the equation (a) is mean annual groundwater level (cm), $NO_3$-N in (b) is mean annual amount of nitrate in 0 – 20 cm (kg N $ha^{-1}$).

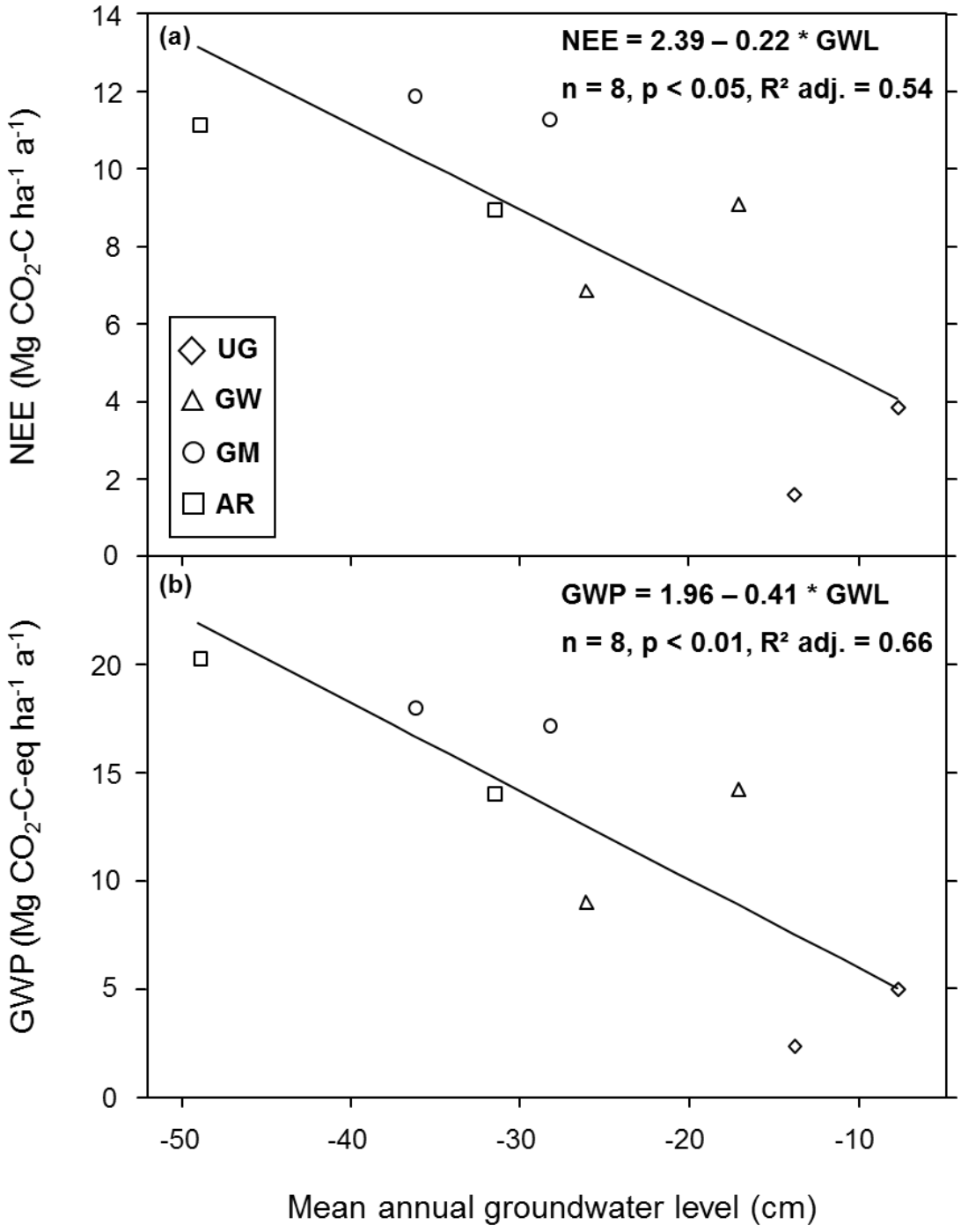


Figure 12. Effect of mean annual groundwater level on net ecosystem exchange (NEE) of $CO_2$ (a) and global warming potential (GWP) of the four study sites (b) during the period April 2012 – March 2014. GWL in the equations is mean annual groundwater level (cm).