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# High methane emissions dominate annual greenhouse gas balances 30 years after bog rewetting

M. Vanselow-Algan<sup>1</sup>, S. R. Schmidt<sup>2,\*</sup>, M. Greven<sup>1</sup>, C. Fiencke<sup>1</sup>, L. Kutzbach<sup>1</sup>,  
and E.-M. Pfeiffer<sup>1</sup>

<sup>1</sup>University of Hamburg, Center for Earth System Research and Sustainability, Institute of Soil Science, Hamburg, Germany

<sup>2</sup>University of Hamburg, Biocenter Klein Flottbek, Applied Plant Ecology, Hamburg, Germany  
\* now at: University of Münster, Institute of Landscape Ecology, Münster, Germany

Received: 16 December 2014 – Accepted: 14 January 2015 – Published: 10 February 2015

Correspondence to: M. Vanselow-Algan (marion.vanselow-algan@uni-hamburg.de)

Published by Copernicus Publications on behalf of the European Geosciences Union.

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## Abstract

Natural peatlands are important carbon sinks and sources of methane ( $\text{CH}_4$ ). In contrast, drained peatlands turn from a carbon sink to a carbon source and potentially emit nitrous oxide ( $\text{N}_2\text{O}$ ). Rewetting of peatlands thus implies climate change mitigation. However, data about the time span that is needed for the re-establishment of the carbon sink function by restoration is scarce. We therefore investigated the annual greenhouse gas (GHG) balances of three differently vegetated bog sites 30 years after rewetting. All three vegetation communities turned out to be sources of carbon dioxide ( $\text{CO}_2$ ) ranging between  $0.6 \pm 1.43 \text{ tCO}_2 \text{ ha}^{-2} \text{ yr}^{-1}$  (*Sphagnum*-dominated vegetation) and  $3.09 \pm 3.86 \text{ tCO}_2 \text{ ha}^{-2} \text{ yr}^{-1}$  (vegetation dominated by heath). While accounting for the different global warming potential (GWP) of the three greenhouse gases, the annual GHG balance was calculated. Emissions ranged between 25 and  $53 \text{ tCO}_2\text{-eqha}^{-1} \text{ yr}^{-1}$  and were dominated by large emissions of  $\text{CH}_4$  (22 up to  $51 \text{ tCO}_2\text{-eqha}^{-1} \text{ yr}^{-1}$ ), while highest rates were found at purple moor grass (*Molinia caerulea*) stands. These are to our knowledge the highest  $\text{CH}_4$  emissions so far reported for bog ecosystems in temperate Europe. As the restored area was subject to large fluctuations in water table, we conclude that the high  $\text{CH}_4$  emission rates were caused by a combination of both the temporal inundation of the easily decomposable plant litter of this grass species and the plant-mediated transport through its tissues. In addition, as a result of the land use history, the mixed soil material can serve as an explanation. With regards to the long time span passed since rewetting, we note that the initial increase in  $\text{CH}_4$  emissions due to rewetting as described in the literature is not limited to a short-term period.

## 1 Introduction

Covering only 3 % of the Earth's land surface, peatlands store as much carbon as all terrestrial biomass and twice as much as all global forest biomass (Parish et al., 2008). Today, merely 1 % of the former extent peatlands in north-western Europe has still

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living mire vegetation and accumulates peat (Koster, 2005). An area of 80 million ha of peatlands have been destroyed worldwide, mainly due to drainage for agriculture and forestry, and due to peat mining for fuel and horticulture (Joosten and Clarke, 2002). While growing mires have a cooling effect on the climate by acting as a carbon sink, degraded peatlands are a major and growing source of the greenhouse gases (GHG) carbon dioxide ( $\text{CO}_2$ ) and nitrous oxide ( $\text{N}_2\text{O}$ ) (Joosten et al., 2012).

$\text{CO}_2$  emissions from degraded peatlands are estimated to be equivalent to more than 10 % of the global fossil fuel emissions (Parish et al., 2008). Even in an industrialized country, like Germany, the estimated peatland GHG exchange accounts for 2.3–4.5 % of the anthropogenic emissions (Drösler et al., 2008). Therefore, restoration of peatlands implies not only a recovery of ecosystem functions and biodiversity, but also climate change mitigation (Drösler et al., 2009). Rewetting of drained peat soils as climate change mitigation measure presents a new challenge (Erwin, 2009; Couwenberg, 2009): in Germany, a potential reduction of 35 million t carbon dioxide equivalents per year is possible by peatland restoration, which is a cost-effective mitigation strategy (Joosten, 2006; Drösler et al., 2009; Freibauer et al., 2009). On the other hand, rewetting of peatlands leads to increased methane emissions by both reduced methane oxidation and increased methane production (Komulainen et al., 1998; Tuittila et al., 2000; Waddington and Day, 2007; Wilson et al., 2009; Cooper et al., 2014). Flooding of peat soils for restoration should be avoided as inundation leads to huge  $\text{CH}_4$  emissions especially if fresh plant litter is available (Augustin and Joosten, 2007; Drösler et al., 2008; Hahn-Schöfl et al., 2011). Although these high  $\text{CH}_4$  emissions are generally assumed to be a transient phenomenon of limited duration (Morison, 2012; Joosten et al., 2012; Artz et al., 2013; Cooper et al., 2014), no information exists on what time span after rewetting this initial phase covers (Augustin and Joosten, 2007). Nevertheless, the term “short-term” is often used in grey-literature (Trepel, 2008; Natural England (ed), 2010; Morison, 2012). Moreover, there is high uncertainty about the magnitude of  $\text{CH}_4$  fluxes in peatlands (Joabsson et al., 1999). However, it is generally agreed that vascular plants stimulate  $\text{CH}_4$  emissions, by

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allowing the gas to bypass the oxygenated upper soil layer moving through the plant tissues (Joabsson et al., 1999; Kutzbach et al., 2004; Lai, 2009). This plant-mediated transport can represent between 30 % and almost 100 % of the total methane flux (Bhullar et al., 2013). Plants that provide such a shortcut between the root zone and the atmosphere are referred to as shunt-species (Joosten et al., 2012). Many studies linked the increase of  $\text{CH}_4$  emissions post-restoration to the colonization of the rewetted area by *Eriophorum* species (Tuittila et al., 2000; Waddington and Day, 2007; Green and Baird, 2012; Cooper et al., 2014).

In the future, restoration and management of peatlands will be more complex and challenging due to climate change (Erwin, 2009; Grand-Clement et al., 2013). Thus, a profound knowledge of peat soil processes will help in adaptation of management practices and in turn in stabilization of ecosystem functions of peatlands against new environmental threats. However, data about the C-balance of restored and abandoned peatlands is scarce on a national and global scale and is urgently needed (Yli-Petäys et al., 2007; Artz et al., 2013). As most studies are focusing on the boreal region or were conducted mainly at recently rewetted sites (Beyer and Höper, 2014), research on the mid and long term effect of rewetting is essential, especially in temperate peatlands.

We studied the  $\text{CO}_2$  and GHG balances of three different plant communities which established at a regenerating mined area and of the active peat extraction site within the same ombrotrophic peatland in northern Germany. Our main goal was to investigate if restoration succeeded in creating a  $\text{CO}_2$  sink and to estimate the GHG mitigation potential of restoring the industrial peat extraction site after abandonment. We hypothesized that the restored site acts as a  $\text{CO}_2$  sink with varying magnitude depending on plant community. Likewise, we expected that higher  $\text{CH}_4$  emissions are related to high abundance of vascular plant species and thus vary between the differing plant communities. On the other hand, no significant  $\text{N}_2\text{O}$  fluxes were expected on the vegetated sites. Testing these hypotheses is expected to provide implications for site-specific management actions, because peat mining of the study area will be ceased in a few years.

## 2 Material and methods

### 2.1 Study area Himmelmoor

The Himmelmoor (Quickborn, Germany;  $53^{\circ}44'20''$  N,  $9^{\circ}50'58''$  E) is located approximately 20 km north-west of Hamburg in Schleswig-Holstein. The mean precipitation sum measured at the climate station in Quickborn is 838 mm per year, and the mean air temperature is  $9.0^{\circ}\text{C}$  (long-term average from 1981 to 2010, data source: Deutscher Wetterdienst). Climatically, the study year 2011 was with 821 mm and  $9.6^{\circ}\text{C}$  very close to this long-term average. We therefore consider our measurements as representative for the present climatic conditions.

With an extent of about  $6\text{ km}^2$  the Himmelmoor is one of the largest raised bogs in Schleswig-Holstein. Peat formation by terrestrialization started after the last ice age  $10\,020 \pm 100$  years before present and the total peat thickness reached a maximum of 10 m (Pfeiffer, 1998; Grube et al., 2010). The original ombrotrophic peat bog was altered by peat drainage and cutting since the 18th century. Commercial peat mining started in 1871 (Averdieck, 1957) and persists until now. According to the peat company a volume of  $38\,000\text{ m}^3$  peat was harvested in the study year 2011 on an area of 70 ha of the former 130 ha extraction site (K. Czerwonka, personal communication, 2013). Peat mining will be ceased in 2016 due to exhaustion of usable peat resources, and restoration will take place.

### 2.2 Measurement sites

Measurements were done on the bare and deeply drained active peat extraction site and on a restored and vegetated part in the north-western Himmelmoor. All soils are classified as *Fibric Ombric Histosol*, while for the peat extraction site the suffix qualifier drainic applies additionally (IUSS, 2006). The restored site was formerly used by hand block-cutting for fuel. Before cutting the drained black peat, the upper rooted layer called *Bunkerde* (Poschlod, 1988) was removed and used for filling up cut-over areas

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as described in Koster (2005). In the 1960s the site was abandoned and vegetation development started mainly from whole plants or generative and vegetative propagules in the *Bunkerde*. Restoration began in the 1980s with drainage-blocking and repeated cutting of birches (last event: 2008) in order to raise the water table. Today, the strips between former drainage ditches (distance: 30–45 m) show differences in plant species composition. Three strips with differing vegetation, typical for rewetted peatlands, were chosen and named after the most prominent plant species or groups later referred to as “heath”, “*Sphagnum*” or “purple moor grass” site. According to the decimal scale of Londo (1976), plotwise analyzes of vascular plant species were done in the study year 2011 during summer time, providing vegetation coverage and abundance to confirm the visual classification of these sites. The moss cover was determined using a pointintercept sampling as described in Jonasson (1988), with the following categories: *Sphagnum* mosses, other mosses, liverworts, and bare peat.

GHG-fluxes were measured at each of the four sites in four replicates (plots). Therefore, PVC frames (60 cm × 60 cm) with a soil insertion depth of about 50 cm were permanently installed blockwise. Positions were selected based on vegetation and microtopography to assure that they represent the whole site. All plots of the vegetated site were equipped with wooden boardwalks to minimize soil gas disturbances during flux measurements. Boardwalks were oriented northwards of the plots to avoid shading. Several micrometeorological variables were continuously monitored at a central point of the restored site, including air temperature and air pressure, photosynthetically active radiation (PAR), wind speed and wind direction (all at 2 m height) and precipitation. Water table depth and soil temperature (10 cm depth) were measured at each site except for the peat extraction site.

### 2.3 Chamber design and flux measurement procedure

GHG flux measurements were done using closed chambers. Their design conforms to the latest recommendations for chamber design made by Pihlatie et al. (2013). CO<sub>2</sub> fluxes were measured with a transparent, climate-controlled chamber system

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connected to an infrared gas analyzer (IRGA, LI-840, LI-COR inc.) as described by Schneider et al. (2011). Additionally, the chamber was equipped with an inside PAR-sensor. Measurements were done for three minutes recording CO<sub>2</sub> concentration, PAR and chamber air temperature every second. To gain a wide spectrum of different light conditions for modeling, the transparent chamber was shaded in two intensities with black gauze (Elsgaard et al., 2012; Görres et al., 2014). After the first measurement with the transparent chamber, a second measurement was performed while shading the chamber with one layer of gauze (PAR approx. 50 %) and a third one with two layers (PAR approx 30 %). Subsequently the chamber was darkened in a fourth measurement with an opaque cover (PAR = 0) to estimate ecosystem respiration (Reco). Between each measurement the chamber was removed and ventilated to obtain ambient CO<sub>2</sub> concentrations within the chamber. If vegetation exceeded chamber height, a transparent polycarbonate elongation of 60 cm height was used, which was shaded and darkened correspondently. Measurements were generally conducted between 10 a.m. and 2 p.m. when PAR reached the maximum and measuring order of the sites was randomized. CO<sub>2</sub> flux measurements were performed the whole year round from August 2010 until January 2012. Measurement intervals depended on vegetation growth with higher frequency in summer than in winter (at least biweekly up to twice a week). To compare day and night respiratory fluxes, night time measurements were done three times during the night of 30 to 31 August 2011 (around 9 p.m., midnight and 5 a.m.). CH<sub>4</sub> and N<sub>2</sub>O flux measurements were performed over a one-year period from April 2011 until March 2012. Measurements were carried out every two weeks (except December 2011: once per month) for CH<sub>4</sub> and monthly for N<sub>2</sub>O as first results showed no significant N<sub>2</sub>O fluxes. CH<sub>4</sub> was measured at all four replicate plots, N<sub>2</sub>O only at three. Night time measurements were performed during the night of 30 to 31 August 2011 around midnight. CH<sub>4</sub> and N<sub>2</sub>O flux measurements were done using aluminum chambers (60 cm × 60 cm × 32 cm; an elongation of 60 cm was used if needed) which were equipped with a fan, a pressure vent, a temperature sensor and a sampling port. Two circular openings (4 cm diameter) at the front site were

open while placing the chamber on the collar and closed afterwards to reduce initial pressure shocks (Schneider et al., 2009). During the closing time of 20 min, six samples were taken from the chamber headspace with 60 mL plastic syringes connected to the sampling port via three-way stopcocks.

5  $\text{CH}_4$  samples were analyzed subsequently in the lab using a gas chromatograph (GC) equipped with a flame ionization detector (HP 5890 Packard Series II). Syringes were directly connected via a loop. Analyses were done within four days after sampling, and each sample was analyzed twice. Two standard gases were used for calibration (1.7 ppm and 200 ppm  $\text{CH}_4$ ) being injected triply before and after samples of three plots.

10  $\text{N}_2\text{O}$  was measured at a GC provided with an electron capture detector (Agilent Technologies 7890A). In the field, a sample volume of 20 mL was injected into an air-filled septum-vial, from where it was then taken with a microliter syringe and injected into the GC. The GC was calibrated daily with three standard gases (0.3, 0.9 and 1.5 ppm  $\text{N}_2\text{O}$ ) being injected triply before measurement. Since the samples were 15 diluted by injecting them into the air-filled vials, standards were treated similar to have the appropriate concentration. As this procedure might cause mistakes and a decrease in accuracy, it was tested with a standard gas. There was no decrease in reproducibility in comparison to a standard gas that was injected directly into the GC: the coefficient of variation was 0.01 in both cases ( $N = 10$ ).

## 20 2.4 Flux calculation

The GHG flux rates were calculated from the change in gas concentration as a function of time during chamber closure. Gross primary production was calculated as the difference between the directly following measurements with transparent and dark chambers, respectively. According to the micrometeorological sign convention, positive values represent fluxes to the atmosphere and negative values uptakes by the 25 ecosystem.

Flux calculation of each single  $\text{CO}_2$  chamber measurement was done with an updated version of the MATLAB<sup>®</sup> routine of Kutzbach et al. (2007) using a power series

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expansion of a non-linear regression as described by Görres et al. (2014). The first and the last 10 s of the 180 data points of each measurement were discarded and the flux rate was calculated at  $t = 10$  s from the remaining 160  $\text{CO}_2$  concentration data points applying a water vapor dilution correction. Each single flux curve was reviewed for abnormalities such as abrupt changes in slope due to e.g. changes in PAR derived from cloud movement. If possible, the flux was recalculated by using only a part of the 160 s interval with constant conditions (minimum 40 s). Flux calculations with outlying residuals were checked for mistakes e.g. in data preparation and were discarded from the dataset if the mistake could not be eliminated. SD of the residuals of most of the data (98 %) was lower than 0.55 ppm and had a mean of  $0.42 \pm 0.06$  ppm, which is remarkably low as the noise of the IRGA is specified to be  $< 1$  ppm. Upward concave flux curves, which are not explainable by diffusion theory were calculated using linear regression as executed by Schneider et al. (2011). It was shown that this procedure achieves more robust and less biased flux estimates (Schäfer, 2012; Görres et al., 2014).

Flux calculation of each  $\text{CH}_4$  and  $\text{N}_2\text{O}$  chamber measurement was done with an updated version of the MATLAB<sup>®</sup> routine of Forbrich et al. (2010). Each single flux curve was reviewed for abnormalities such as ebullition and discarded from dataset if necessary. Linear or non-linear regression was used depending on the model performances according to the Akaike information criterion. As only six concentration measurements were available for flux calculation, we used the Akaike information criterion with small sample correction (AICc), as proposed by Forbrich et al. (2010). According to AICc, the majority of  $\text{CH}_4$  and  $\text{N}_2\text{O}$  fluxes curves was better explained by linear than by exponential regression (74 and 98 % respectively).

## 25 2.5 Flux modeling

Net ecosystem exchange (NEE), ecosystem respiration (Reco) and gross primary production (GPP) were modeled over a complete year (2011). Modeling of GPP and Reco was based on ambient PAR and air temperature, respectively, which were

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measured half-hourly at the meteorological station. Each single plot ( $N = 16$ ) was modeled separately, to analyze the differences between and within the sites. Values of the 4 plots per site were later averaged, thus SDs shown here display the spatial variability of flux estimates. As day and night fluxes of  $\text{CO}_2$ ,  $\text{CH}_4$  and  $\text{N}_2\text{O}$  were not statistically different, modeling was only based on daytime data.

GPP was modeled with a rectangular hyperbolic light response curve as described in Elsgaard (2012) using PAR values from inside the chamber (GPP model 1, Table 1). Applying this curve obtains light saturation points (Pmax) for a certain period of the year (one week up to one month depending on measurement interval and season). Gap filling between these intervals was done using linear interpolation. If model 1 could not explain the data appropriate, a linear model was used instead (GPP Model 2, Table 1). In winter time when GPP ranged around zero, none of the two models could be fitted and mean values were used instead.

To estimate annual Reco fluxes, we used air and soil temperature as explanatory variables and tested two different models (Table 1), using all respiratory flux data of the study year for each plot. Comparing the qualifying parameter  $R^2_{\text{adjusted}}$  of the model results showed that the respiratory fluxes were better explained by air than by soil temperature and that Reco-model 1 achieved better results than Reco-model 2. The annual Reco fluxes of the industrial extraction site were estimated using linear regression, as both models resulted in low  $R^2_{\text{adjusted}}$  values ranging between 0.03 and 0.33.

Methane fluxes did not show a soil temperature or water table depth dependency. Annual fluxes were thus not modeled with the classical model of Saarnio et al. (1997) as intended, but calculated by linear interpolation as well as the annual  $\text{N}_2\text{O}$  flux rates.

## 2.6 Calculation of the GHG budget

For the calculation of the greenhouse gas (GHG) budget, the fluxes of  $\text{CH}_4$  and  $\text{N}_2\text{O}$  were converted in  $\text{CO}_2$  equivalents according to their global warming potentials on a 100 year time scale including climate-carbon feedbacks:  $\text{CH}_4 = 34$  and  $\text{N}_2\text{O} = 298$

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(IPCC, 2013). The C-loss due to peat mining was also estimated as CO<sub>2</sub> equivalents. The calculation was based on the amount of peat harvested in the study year in respect of bulk density and C-content of the peat, measured within the upper three soil horizons. Although not all of the peat harvested in 2011 decomposes to CO<sub>2</sub> within that year, the C-loss was completely added to the balance as it is effectively apparent on site.

### 3 Results

#### 3.1 Water table

The four sites displayed differences in water table depth throughout the year (Fig. 1).

10 The *Sphagnum* site had the highest water table followed by the heath and the purple moor grass site. The lowest water table was recorded at the drained active industrial extraction site with a minimum of 55 cm under the soil surface showing a great range of water table amplitude. Contrary, the variation in the water table depth of the three vegetated sites was much less. Soil surfaces of the rewetted sites were often inundated  
15 especially in the winter half-year.

#### 3.2 Vegetation

The vegetation analysis showed clear differences in species coverage between the three defined sites (Table 2). The heath site was dominated by ericaceous shrubs (49 % cumulative coverage of *E. tetralix*, *C. vulgaris*, *V. oxycoccus* and *A. polifolia*).

20 The *Sphagnum* site had 99 % coverage of *Sphagnum* mosses (mainly *S. cuspidatum* and *S. fimbriatum*) and the purple moor grass site was dominated by the perennial deciduous grass *Molinia caerulea* (67 % coverage). Sites additionally differed in the number of vascular plant species decreasing in the following order: heath > *Sphagnum* > purple moor grass with in total nine, eight and six species, respectively.

### 3.3 NEE model

The results of the modeled ecosystem respiration (Reco) (Fig. 2) and the modeled gross primary production (GPP) were summed up resulting in modeled net ecosystem exchange (NEE) (Fig. 3). The temporal dynamics between the sites were quite similar for the heath and the *Sphagnum* site. However, the *Sphagnum* site displays some more pronounced peaks in photosynthesis. The purple moor grass site has a steep increase in photosynthesis in May and is very productive in summertime, while photosynthesis at the two other vegetated sites begins earlier and with a slower increase. Before the steep increase in photosynthesis at the purple moor grass site, respiratory fluxes predominate and increase from March until May. The industrial extraction site is characterized by comparatively low respiratory fluxes.

### 3.4 CO<sub>2</sub> budget

The annual net ecosystem exchange (NEE) in 2011 was positive at all vegetated sites as well as at the industrial extraction site (Fig. 4), meaning a net CO<sub>2</sub> release from the ecosystem. The highest values of NEE were determined at the active industrial extraction site where ecosystem respiration predominates ( $730 \pm 67 \text{ g m}^2 \text{ yr}^{-1}$ ), followed by heath, purple moor grass and *Sphagnum* with  $308 \pm 386$ ,  $247.36 \pm 330.29$  and  $59.56 \pm 142.58 \text{ g m}^2 \text{ yr}^{-1}$  respectively. These differences in NEE between the sites were closely approximating significance (ANOVA,  $p = 0.055$ ).

### 3.5 CH<sub>4</sub> and N<sub>2</sub>O fluxes

CH<sub>4</sub> emissions showed no seasonal trend and were not dependent on water level or soil temperature (data not shown). However, CH<sub>4</sub> fluxes differed significantly among sites (ANOVA,  $p < 0.00$ ): CH<sub>4</sub> fluxes of the industrial extraction site ranged around zero, while the vegetated sites acted as CH<sub>4</sub> sources. CH<sub>4</sub> emissions increased in the following order (Table 3): heath < *Sphagnum* < purple moor grass

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( $2.02 \pm 0.41$ ,  $3.16 \pm 2.20$  and  $4.71 \pm 3.32 \mu\text{g m}^{-2} \text{s}^{-1}$ , respectively) and increased with increasing coverage of vascular plant species with aerenchyma (sum of *M. caerulea*, *E. vaginatum*, *E. angustifolium* and *Juncus* spec., Fig. 5).  $\text{N}_2\text{O}$  fluxes were significantly different between sites (ANOVA,  $p = 0.002$ ). Significant emission of  $\text{N}_2\text{O}$  were only measured at the industrial extraction site at summer time, resulting in mean annual emissions of  $0.0165 \pm 0.0086 \mu\text{g m}^{-2} \text{s}^{-1}$ .  $\text{N}_2\text{O}$  fluxes of the rewetted sites were very low, ranging around zero and varying between small uptakes and emissions throughout the whole year.

### 3.6 Greenhouse gas balance

$\text{CH}_4$  and  $\text{N}_2\text{O}$  fluxes were calculated into  $\text{CO}_2$  equivalents ( $\text{CO}_2\text{-eq}$ ) and summed up with the  $\text{CO}_2$  fluxes as a GHG budget (Table 4, for a better visualization values are demonstrated in Fig. 6). We found that all sites are significant sources of greenhouse gases. The GHG balance of the rewetted sites is dominated by  $\text{CH}_4$  emissions, accounting for 88–98 % of the total GHG  $\text{CO}_2\text{-eq}$  emissions. The portion of  $\text{N}_2\text{O}$  is negligible there, while it plays with about 21 % of the GHG balance a significant role at the industrial extraction site ( $1.55 \pm 0.81 \text{ tCO}_2\text{-eq ha}^{-1} \text{ yr}^{-1}$ ). The uncertainty of the GHG budget is high due to high spatial variability between the replicates. Nevertheless, differences between the sites were statistically different ( $p < 0.05$ , Kruskal–Wallis one way ANOVA). The highest GHG emissions were detected at the purple moor grass site ( $53.05 \pm 35.72 \text{ tCO}_2\text{-eq ha}^{-1} \text{ yr}^{-1}$ ), whereas the lowest ones were present at the industrial extraction site ( $8.9 \pm 1.1 \text{ tCO}_2\text{-eq ha}^{-1} \text{ yr}^{-1}$ ). However, the GHG balance of the industrial extraction site is not complete without considering the amount of mined peat. Including these C-losses and assuming their release as  $\text{CO}_2$ , the GHG balance of the industrial extraction site was considerably higher than those of the vegetated sites ( $123 \pm 7 \text{ tCO}_2\text{-eq ha}^{-1} \text{ yr}^{-1}$ ).

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## 4 Discussion

The visual differences in plant communities between the three sites were confirmed by analyses of species composition and coverage. Additionally, different life-form types dominate, emphasizing diverse functions and adaptations to the ecosystem: evergreen, 5 ericaceous shrubs (heath site), *Sphagnum* mosses with sedges (*Sphagnum* site) and nearly monospecific stands of the perennial deciduous grass *M. caerulea* (purple moor grass site). We presume the differences in vegetation composition to be a result of the different water table levels. As the lowest water table position was detected at the purple 10 moor grass site, the interaction between the relatively drier surface and the appearance of the non-typical bog plant results in significant differences in GHG-fluxes compared to the other two vegetated sites. Here, the highest  $\text{CH}_4$  fluxes were measured, as well as the highest gross primary production and highest maximum photosynthetic activity. The 15 fast growth of *M. caerulea* leaves and their autumn senescence can explain the annual dynamics of the modeled NEE at the purple moor grass site. This grass performs no photosynthesis in winter time besides from green basal internodes (Jefferies, 1915). Thus, photosynthesis steeply increases with the springtime growth and abruptly stops 20 with the dieback of its leaves in fall. By contrast, photosynthesis was detected in winter time at the *Sphagnum* and the heath site as they were dominated by evergreen plants.

All three vegetation communities established at the restored study site turned out 25 to be sources of  $\text{CO}_2$ . Thus, restoration failed to re-establish the  $\text{CO}_2$  sink function after 30 years as hypothesized. However, the *Sphagnum* site had the lowest  $\text{CO}_2$  emissions and ranges, with its uncertainty associated to the spatial variability, close to a  $\text{CO}_2$  neutral status. These results are concordant with the findings of Samaritani et al. (2010) who showed that a *Sphagnum*-dominated European mid-latitude cut-over bog, was a  $\text{CO}_2$  source 29 years after rewetting, but older regenerating sites (42 and 51 years respectively) were taking up  $\text{CO}_2$ . The magnitude of the  $\text{CO}_2$  balances is consistent with the analyses of 53 studies evaluated by Couwenberg et al. (2008), showing that temperate European peatlands with similar water table positions emit

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less than  $3\text{tCO}_2\text{ha}^{-1}\text{yr}^{-1}$  (utilisation and restoration status not mentioned). The net  $\text{CO}_2$  emissions measured at the industrial extraction site are higher compared to the restored sites, as the soil is deeply drained and no photosynthesis takes place. The annual  $\text{CO}_2$  emission of  $7.3\text{tCO}_2\text{ha}^{-1}\text{yr}^{-1}$  is ranging in the dimension of 11 reported studies from European drained peat cut sites with a median of 2 and a maximum of  $13\text{tCO}_2\text{ha}^{-1}\text{yr}^{-1}$  (Drösler et al., 2008). Rewetting of the industrial extraction site provides a  $\text{CO}_2$  mitigation potential between 422 and  $671\text{gCO}_2\text{m}^{-2}\text{yr}^{-1}$  (peat extraction not included) calculated from the  $\text{CO}_2$  budgets of the three rewetted sites. This is in total 295 up to  $470\text{tCO}_2\text{yr}^{-1}$  for the whole site (70 ha), depending on which of the three vegetation communities establishes. However, restoring a cut-over peatland requires special techniques as most *Sphagnum* species cannot re-establish spontaneously and re-vegetation is generally slow due to no viable propagules and unfavorable conditions for plant growth and seed germination (Quinty and Rochefort, 2003; Triisberg et al., 2011; D'Astous et al., 2013).

The GHG budget of the restored site is dominated by  $\text{CH}_4$  fluxes (up to 99 %). While the magnitude of the modeled  $\text{CO}_2$  balances is similar to data reported in other studies as described above, the magnitude of the  $\text{CH}_4$  fluxes is very high. To our knowledge, the maximum rate reported for a European bog is approximately  $17\text{tCO}_2\text{-eqha}^{-1}\text{yr}^{-1}$ , which was measured in the Bavarian Alpine foreland on a natural bog site with *Sphagnum-Scheuchzeria palustris* hollows (Drösler, 2005). In the present study, we found annual  $\text{CH}_4$  fluxes of 21, 34 and  $50\text{tCO}_2\text{-eqha}^{-1}\text{yr}^{-1}$ , respectively. We hypothesize that the large fluctuations in water level, which are typical for degraded peatlands (Schouwenaars, 1993; Tuittila et al., 1999), might explain these high  $\text{CH}_4$  emissions as fresh belowground or aboveground plant litter is inundated episodically and undergoes anoxic fermentation (Augustin and Joosten, 2007; Paul and Alewell, 2013). This applies especially for the purple moor grass site although it features the lowest water level of the three vegetated sites: here,  $\text{CH}_4$  fluxes were higher than at the *Sphagnum* and the heath site (2.3 and 1.5 times, respectively). We suggest that this is due to the large amounts of litter produced by *M. caerulea*, which is easily

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decomposable in comparison to other plants present at the study site (van Breemen, 1998). Like we found here, Bohdalkova et al. (2013) observed highest  $\text{CH}_4$  emissions at the site with the lowest water level and suggested this to be the result of the invasion of easily degradable vascular plants. Likewise, Hahn-Schöfl et al. (2011) explained 5 extremely high  $\text{CH}_4$  emissions in a German fen with the inundation of reed canary grass (*Phalaris arundinacea*). Another likely explanation for the huge  $\text{CH}_4$  fluxes from the purple moor grass site is the presence of aerenchyma (Jaiswal et al., 2000), by which *M. caerulea* can act as gas conduit. It hence allows  $\text{CH}_4$  to bypass the oxidized surface soil, and consequently reduces  $\text{CH}_4$  oxidation. Although the flux of  $\text{CH}_4$  through 10 the plant tissues is lower in *M. caerulea* compared to *E. angustifolium* (Bhullar et al., 2013), the chimney effect might be increased by the high cover of this grass species at the purple moor grass site. Thus, it was shown, that  $\text{CH}_4$  emissions increase with increasing coverage of shunt species. The composition of plant communities is therefore the most important driver for  $\text{CH}_4$  fluxes at the study sites, as reported by 15 other authors (Samaritani et al., 2010; Couwenberg et al., 2011; Bohdalkova et al., 2013).

Irrespective of the abundance of shunt species,  $\text{CH}_4$  emissions from the rewetted study site are generally very high. We therefore assume that the former land use history forms fundamental condition for methane production. As mentioned above, the 20 upper vegetated soil layer (*Bunkerde*) was used for filling up the mined area. We found differently decomposed soil material in layers or lens-shaped zones in the soil profiles of all rewetted sites (data not shown). We are hypothesizing that the peat material, which was exposed to oxygen enhances methanogenic degradation of the organic matter by the initiated decay process. Additionally, labile organic matter in the form of 25 aboveground plant biomass of the primary vegetation was buried and thus might serve as substrate for  $\text{CH}_4$  production. A similar effect was found by Cooper et al. (2014), who identified former drainage ditches fulfilled with heather bales and colonized by aerenchymatous species as hotspots for  $\text{CH}_4$  emissions.

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These findings are important for a large area of exploited peatlands, because the peat extraction method applied here was the traditional way of peat cutting (Koster, 2005). It is therefore essential that more attention is paid to monitoring and management of areas with a hand-cutting history. In terms of  $\text{CH}_4$  mitigation and climate protection an active reduction of *M. caerulea* stands by e.g. sod cutting or grazing could help to establish target species communities instead (Jacquemart et al., 2003; Reid et al., 2009; Keddy, 2010; Meuser, 2012), and hence reduce  $\text{CH}_4$  emissions at the study site. Avoiding water table fluctuations is additionally important to impede inundation of fresh plant litter as a substrate for methanogenesis (Hahn-Schöfl et al., 2011). The installation of overflows as management measure could stabilize the water table level (Maitland and Morgan, 1997; Quinty and Rochefort, 2003).

## 5 Conclusion

We found large differences in GHG fluxes between the three vegetation communities. Differentiation between plant communities for estimation and upscaling of GHG budgets is therefore imperative. The particular land use history of this degenerated peat site, in combination with water tables fluctuations and a high cover of aerenchymatous plants seem to form favorable conditions for both, high methanogenesis and efficient soil atmosphere  $\text{CH}_4$  transport. This results in extremely high  $\text{CH}_4$  emissions and high GHG budgets in comparison to other bog ecosystems in Europe. The annual GHG emissions are at least twice as high as the rates of the industrial extraction site, although 30 years passed since rewetting. We therefore state that the initial increase in  $\text{CH}_4$  emissions due to rewetting is not limited to a short-term period, as described in literature. We are raising the question if the emergence of high methane fluxes is, on the contrary, permanent in severely damaged rewetted peatlands, as it is not possible to fully reestablish ecosystem functions.

*Acknowledgements.* We wish to acknowledge the support of the peat company manager in the Himmelmoor Klaus Czerwonka, and the committed assistance in field and lab of

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Birgit Grabellus, Mathias Schwarzer, Stephanie Langer and Kira Kalinski. We appreciate the technical support from Christian Wille and Peter Schreiber during the planning and preparation stages of this study. This work was carried out within the research project KLIMZUG-Nord, which was funded by the German Federal Ministry of Education and Research (grant number 5 01LR0805D). L. Kutzbach was supported through the Cluster of Excellence “CliSAP” (EXC177) at the University of Hamburg funded through the German Research Foundation.

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**Table 1.** Two model approaches for gross primary production (GPP) as a function of photosynthetically active radiance (PAR) and temperature driven ecosystem respiration (Reco) models, where  $t$  = air/soil temperature, respectively and  $a$  and  $b$  are fitting parameter.

Model title	Model formula	Remarks
GPP 1	$GPP = \frac{P_{max}\alpha PAR}{P_{max} + \alpha PAR}$	Rectangular hyperbolic function (Schäfer, 2012; Elsgaard et al., 2012). $P_{max}$ = maximum potential photosynthetic rate, $\alpha$ = initial light response efficiency.
GPP 2	$GPP = a + bPAR$	Linear model.
Reco 1	$Reco = \frac{a}{1 + be^{-kt}}$	Simple logistic function (Richards, 1959; Rodeghiero and Cescatti, 2005; Schäfer, 2012).
Reco 2	$Reco = ae^{bt}$	Two parameter exponential function (Schneider et al., 2009; Schäfer, 2012).

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**Table 2.** Average coverage of vascular plant species according to Londo (1976) at three rewetted bog sites in 2011. Coverage of the moss layer and bare peat areas were estimated using point intercept sampling (Jonasson, 1988). Values refer to means of 7–8 replicates  $\pm$  SD. Numbers in bold print indicate eponymous species for site denotation. *Sphagnum* species were mainly *S. cuspidatum* and *S. fimbriatum*.

Plant species coverage (%)	Heath	<i>Sphagnum</i>	Purple moor grass
<i>Eriophorum angustifolium</i>	$16.9 \pm 8.4$	$45.7 \pm 29.2$	$3.4 \pm 3.9$
<i>Molinia caerulea</i>	$1.3 \pm 1.8$	$18.4 \pm 21.2$	<b><math>66.9 \pm 15.6</math></b>
<i>Erica tetralix</i>	<b><math>30.6 \pm 19.0</math></b>	$5.7 \pm 8.4$	–
<i>Calluna vulgaris</i>	$14.4 \pm 18.0$	$0.9 \pm 1.9$	$1.4 \pm 3.5$
<i>Eriophorum vaginatum</i>	$5.1 \pm 5.1$	$2.4 \pm 3.8$	$0.1 \pm 0.4$
<i>Vaccinium oxycoccus</i>	$1.6 \pm 1.7$	$0.7 \pm 1.9$	–
<i>Andromeda polifolia</i>	$2.5 \pm 3.8$	–	–
<i>Betula pubescens</i>	$2.0 \pm 3.3$	$1.4 \pm 3.0$	$1.4 \pm 1.6$
<i>Drosera rotundifolia</i>	$0.3 \pm 0.5$	$0.3 \pm 0.5$	–
<i>Juncus</i> spec.	–	–	$0.3 \pm 0.5$
<i>Sphagnum</i> mosses	$23.6 \pm 35.9$	<b><math>98.9 \pm 3.0</math></b>	$17.7 \pm 14.8$
Liverworts	$33.0 \pm 37.9$	–	$13.2 \pm 18.0$
Other mooses	$39.6 \pm 43.5$	$1.1 \pm 3.0$	$5.6 \pm 10.6$
Bare peat	$2.4 \pm 6.9$	–	$22.2 \pm 15.1$

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**Table 3.** Mean annual  $\text{CH}_4$  and  $\text{N}_2\text{O}$  fluxes  $\pm$  SD at four different bog sites. Values refer to means of 4 replicates  $\pm$  SD.

Site	$\text{CH}_4$ ( $\mu\text{gm}^{-2} \text{s}^{-1}$ )	$\text{N}_2\text{O}$ ( $\mu\text{gm}^{-2} \text{s}^{-1}$ )
Heath	$2.02 \pm 0.41$	$-0.0008 \pm 0.0038$
<i>Sphagnum</i>	$3.16 \pm 2.20$	$0.0017 \pm 0.0021$
Purple moor grass	$4.71 \pm 3.32$	$-0.0013 \pm 0.0038$
Industrial extraction	$0.01 \pm 0.02$	$0.0165 \pm 0.0086$

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**Table 4.** Greenhouse gas (GHG) budgets of four bog sites. The GHG budget of the active industrial extraction site is composed of measured fluxes on site and the C-loss due to peat mining in 2011, assuming its emission as CO<sub>2</sub>. Values represent means of four (CO<sub>2</sub> and CH<sub>4</sub>) or three (N<sub>2</sub>O) replicates ± SD.

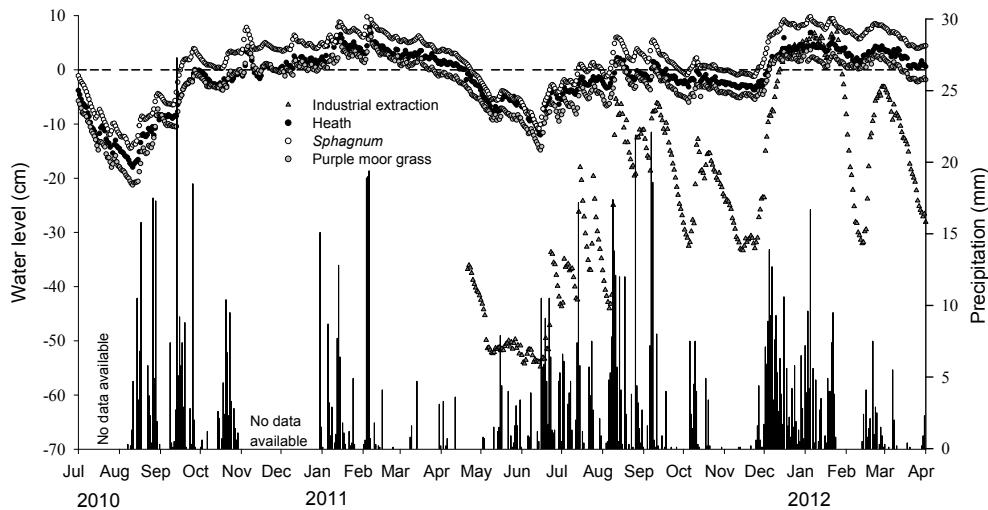
Site	CO <sub>2</sub>	CH <sub>4</sub>	N <sub>2</sub> O (tCO <sub>2</sub> -eqha <sup>-1</sup> yr <sup>-1</sup> )	Sum
Heath	3.09 ± 3.86	21.61 ± 4.44	-0.07 ± 0.36	24.63 ± 5.89
<i>Sphagnum</i>	0.60 ± 1.43	33.87 ± 23.63	0.16 ± 0.20	34.62 ± 23.68
Purple moor grass	2.47 ± 3.30	50.45 ± 35.56	0.12 ± 0.36	53.05 ± 35.72
Industrial extraction				
on site	7.30 ± 0.67*	0.07 ± 0.22	1.55 ± 0.81	8.93 ± 1.07
peat mining				114.02 ± 6.70
total				122.95 ± 6.78

\* only 3 replicates

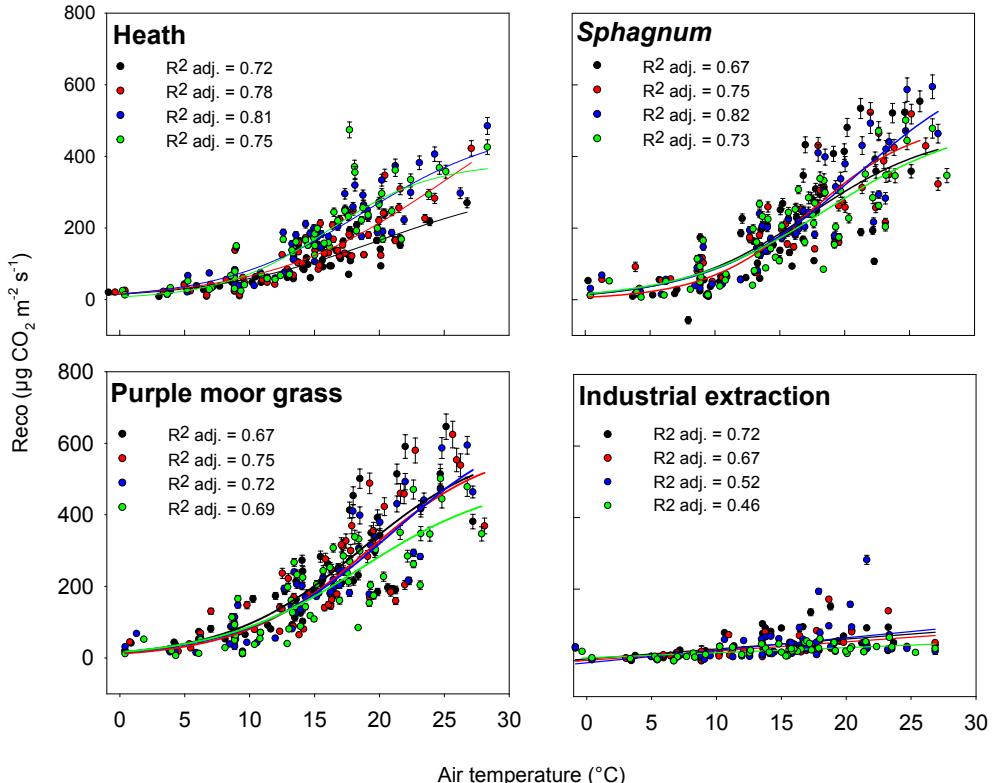
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**Figure 1.** Mean daily water table of four different bog sites and daily precipitation sum over the whole measurement period from July 2010 until April 2012. No data available for the industrial extraction site before April 2011. Negative values indicate water levels below soil surface and positive values indicate inundation.



**Figure 2.** Measured ecosystem respiration (Reco) as a function of air temperature. Points indicate fluxes  $\pm$  standard error of the flux calculation. The replicate plots of each site are shown in different colors. Lines represent the fits to raw data using Reco model 1. For the industrial extraction site a linear regression was used. The goodness-of-fit is indicated ( $R^2_{\text{adjusted}}$ ).

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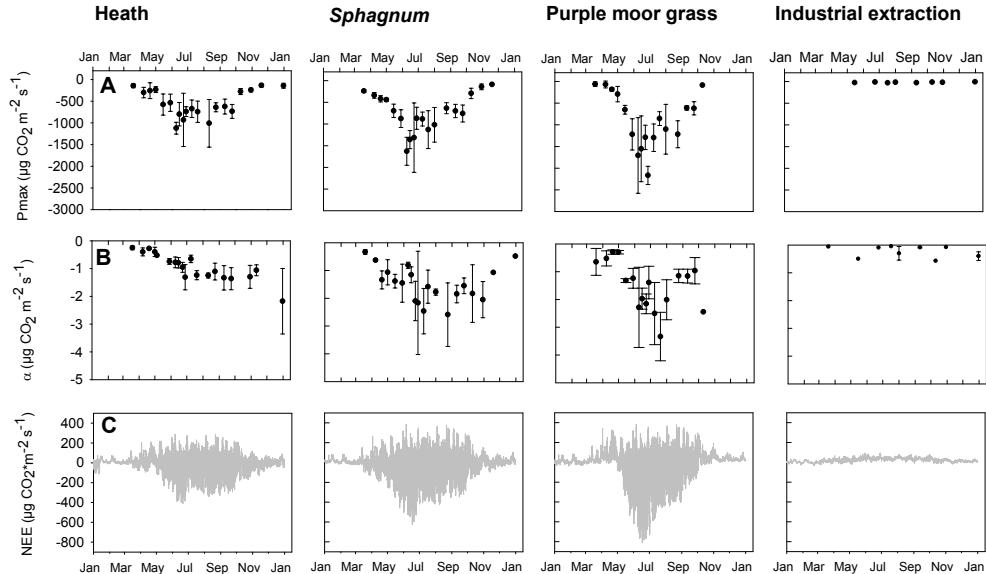
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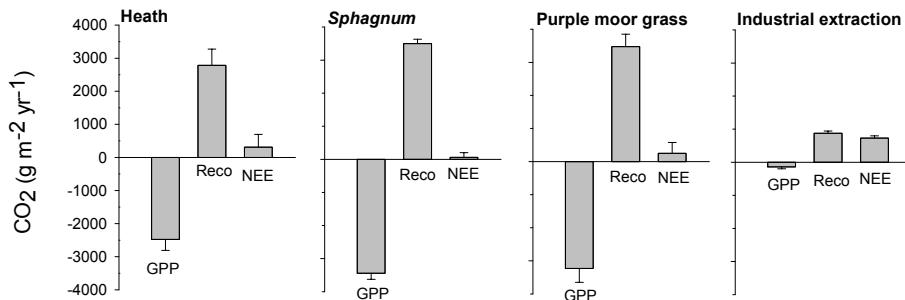


**Figure 3.** Model parameters for gross primary production (GPP model 1) in 2011: **(a)** maximum photosynthetic activity ( $P_{\text{max}}$ ), **(b)** values for the initial light response efficiency ( $\alpha$ ). Points indicate means of four replicate plots  $\pm$  SD, **(c)** mean modeled net ecosystem exchange (NEE) in 2011. Positive values represent emissions and negative values uptakes.

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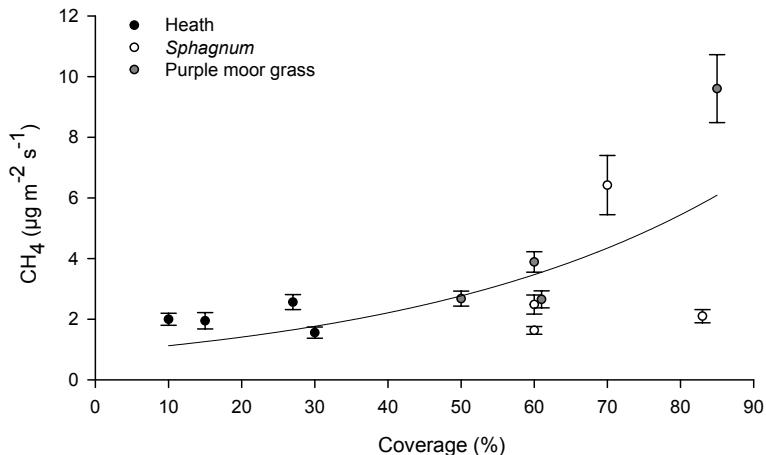
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**Figure 4.** Annual  $\text{CO}_2$  budgets of four different bog sites in 2011. GPP: gross primary production, Reco: ecosystem respiration and NEE: net ecosystem exchange. Values represent means of the model results of four replicates of each site  $\pm$  SD. Positive values represent emissions and negative values uptakes.

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**Figure 5.** Dependency of  $\text{CH}_4$  emissions from the coverage of aerenchymatous plant species (sum of *M. caerulea*, *E. vaginatum*, *E. angustifolium* and *Juncus* spec.) at three different vegetation communities in the studied bog. Shown are the mean annual fluxes per plot  $\pm$  SD and the exponential regression line ( $R^2 = 0.39$ ).

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GHG budget ( $\text{t CO}_2\text{-eq ha}^{-1} \text{yr}^{-1}$ )

Category	$\text{CO}_2$	$\text{CH}_4$	$\text{N}_2\text{O}$	Peat mining (C-export)
Heath	~3	~22	~1	~0
Sphagnum	~3	~35	~0	~58
Purple moor grass	~2	~53	~0	~88
Industrial extraction	~9	~0	~10	~122

**Figure 6.** Greenhouse gas budget in CO<sub>2</sub> equivalents of four bog sites 2011. Bars represent means of four (CO<sub>2</sub> and CH<sub>4</sub>) or three (N<sub>2</sub>O and CO<sub>2</sub> of the industrial extraction site) plots per site  $\pm$  SD. The C-export by peat mining was calculated into CO<sub>2</sub> emissions.