High N_2O and CO_2 emissions from bare peat dams reduce the climate mitigation potential of bog rewetting practices

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SUMMARY

The rewetting of drained peatlands is currently a common practice for re-establishing near-natural hydrological conditions and for reducing peatland greenhouse gas (GHG) emissions, especially of carbon dioxide (CO_2) and nitrous oxide (N_2O), which are enhanced under peatland drainage and extraction. In the originally bare and drained peatland Himmelmoor (Quickborn, Germany), the rewetting process started stepwise in 2004 by blocking drainage ditches with peat as well as by creating polders surrounded by peat dams. In this research we examined differences in CO_2, methane (CH_4) and N_2O emissions between a flooded (FL) area, a bare peat dam (PD) area and an abandoned (but still drained) extraction (E) area during a period in 2012 and during a period of two years in 2014–2016. The results showed that all study areas were GHG sources, although large differences were identified between the different sites. Winter CO_2 emissions from all sites (FL, PD, E) were within the range previously reported for rewetted peatlands, but summer CO_2 emissions from PD (1–20 µmol m^{-2} s^{-1}) strongly exceeded the reported average range for similar surfaces. Very low and irregular CH_4 fluxes were detected at both PD and FL, ranging from -6 to 24 nmol m^{-2} s^{-1} at PD and from -13 to 49 nmol m^{-2} s^{-1} at FL. In comparison to other peatlands, the observed N_2O emissions were high, especially at the PD sites with maximum daily means of 23 nmol m^{-2} s^{-1} in the summer of 2012. In general, the flooded excavation sites (FL) showed lower GHG emissions than the not-rewetted excavation area (E). Also, despite the relatively small coverage of the peat dams (PD), these areas showed a larger total GHG emission than the E and FL sites. This negative effect of peat dams during the first years after flooding could be mitigated by stimulating their colonisation by moss or heath vegetation, which reduces the soil N pool and thus can be expected to reduce N_2O fluxes from the peat dams.

KEY WORDS: greenhouse gas emissions, ombrotrophic mire, peatland, restoration

INTRODUCTION

Despite the fact that peatlands cover only 3 % of the terrestrial surface, they play a key role in global biogeochemical cycles (Gorham 1991, Joosten & Clarke 2002, Vasander *et al.* 2003, Charman *et al.* 2013). Peatlands are characterised by the accumulation of organic soil material consisting of dead plant remains. Globally, peatlands store around 455–550 Gt of carbon, which is equivalent to twice that stored in the world's forest biomass (Kivinen & Pakarinen 1981, Gorham 1991, Bridgham *et al.* 2006, Parish *et al.* 2008). In Germany, peatlands cover 13,000 km^2, which is approximately 4 % of the land area, and are estimated to contain 422 Tg C (Joosten & Clarke 2002, Byrne *et al.* 2004). Of these peatlands, more than 9,300 km^2 have been drained for agricultural use.

Peat extraction and peatland drainage influence biogeochemical processes and can change the amount of exported carbon and nitrogen. Lowering the peatland water table creates favourable conditions for peat oxidation and mineralisation and can turn peatlands from net carbon sinks to net carbon sources. Especially, the CO_2-equivalent emissions of the important greenhouse gases (GHG) carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O), which globally account for 2–3 Gt per year (Joosten & Couwenberg 2009), are known to increase (CO_2 and N_2O) or decrease (CH_4) as a result of drainage (Couwenberg 2011). However, it is difficult to predict changes in peatland GHG emissions under drainage since peatland emissions are reported to be highly variable in space and time, and are dependent on many factors such as soil water content and water table position, soil temperature, plant community structure, redox potential, and the chemical recalcitrance of peat tissue (Yavitt *et al.* 1997, Blodau 2002, Blodau *et al.* 2004, Glatzel *et al.* 2004).

Currently, rewetting and flooding of drained peatlands are common practices for improving hydrological conditions and for reducing peatland

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GHG emissions. The rewetting projects in Germany have been the largest carried out in Europe during recent decades (Zerbe & Wiegleb 2009). Peatland restoration commonly takes place in two stages. The first is the re-establishment of a high water table level (rewetting, flooding) and the second is the recolonisation of important peat-forming *Sphagnum* species (Holden *et al.* 2004, Landry & Rochefort 2012). Currently, there is no universal restoration technique for re-establishing the water table level, as the conditions vary widely. Depending on the drainage duration and the restoration objective, the water level can be raised and stabilised using different methods including: blocking of drainage ditches (e.g. by filling or damming them with wet humified peat or sawdust); installing regulation devices; or building continuous dams (bunds) of peat, wood or plastic around the rewetting site. Peat is the commonest and most cost-effective material for dams, especially if the peatland has a shallow slope meaning many obstructions are required. The different restoration methods lead to changes in hydrology, oxygen availability and soil redox potential, and possibly to the re-establishment of mire-typical soil microbiological and vegetation communities (Weltzin *et al.* 2000, Holden *et al.* 2004). In the context of climate change mitigation research, it is important to determine the impact of these management practices. For this reason, the effects of rewetting of drained peatlands on local GHG emissions has been investigated and monitored in different studies (Christen *et al.* 2016, Jordan *et al.* 2016, Wilson *et al.* 2016). For example, in a German degraded fen, Koebsch *et al.* (2013, 2015) reported that vascular plant communities which were not adapted to flooded conditions suffered, causing a reduction in CO_2 uptake and emission, and led to an increase in CH_4 emissions during the first year of flooding. Similar findings have been reported in other previous studies, where rewetted peatlands have been found to have lower CO_2 emissions (1 to 28 %) and significantly larger CH_4 emissions (78 %) than their drained counterparts (Best & Jacobs 1997, Karki *et al.* 2016, Järveoja *et al.* 2016). Although only relatively few studies have investigated the effect of rewetting on N_2O emissions, research suggests that restoration processes cause an overall reduction in N_2O emissions (Jordan *et al.* 2016, Wilson *et al.* 2016); for example, Davidsson *et al.* (2002) showed that N_2O production was limited by anaerobic (flooded) conditions.

Despite numerous studies on the effect of rewetting practices on GHG emissions (Beyer & Höper 2015, Günther *et al.* 2015), there are still many aspects that require further research. For example, the overall impact of rewetting on large peat extraction areas, especially with extensive peat dams, is unknown. To our knowledge, there are no published studies that have simultaneously measured the emissions of all three GHGs from peat dams.

The goal of this study is to investigate the range, the seasonal variability and the meteorological controls of CO_2, CH_4 and N_2O fluxes from flooded areas (FL), dry bare peat dams (PD) and an abandoned drained extraction area (E) (as reference), all located within the ombrotrophic peatland Himmelmoor in Northern Germany. We hypothesised that: i) the bare peat dam is a CO_2 and N_2O source under oxic conditions, with a variable magnitude depending on the water table level (WTL); ii) no significant CH_4 emissions will be found on sites with dry topsoil (PD and E), in contrast to flooded sites; and iii) the flooded area (FL) will act as a significantly smaller CO_2 source than the abandoned extraction area (E), as a result of reduced peat mineralisation. Validation of these hypotheses is necessary for assessing the suitability of commonly applied peat rewetting and conservation management techniques.

METHODS

Study area

The study was conducted in the peatland Himmelmoor (53° 44′ 20″ N, 9° 50′ 58″ E), a raised ombrogenous bog situated north-west of Hamburg in northern Germany, near Quickborn, Schleswig-Holstein. The region has a temperate climate with warm summers, classified as Cfb according to the Köppen system (Kottek *et al.* 2006), with an annual mean precipitation of 838 mm and an average temperature of + 9 °C for the reference period 1981–2010 (DWD 2016). Peat accumulation is estimated to have begun after the last ice age, 10,020 ± 100 years before present (Pfeiffer & Becker-Heidmann 1996). The peat-covered area accounts for approximately 6 km² (Vanselow-Algan 2014) and the total thickness of the peat accumulation can reach up to 10 m. (Pfeiffer 1998, Vanselow-Algan 2014). The extraction of peat for fuel in Himmelmoor started in the 1780s and continued until 1968. Since 1918, peat has been excavated for horticulture by the peat factory Quickborn (Torfwerk Enfeld Carl Hornung Werk Quickborn), and at present a layer of peat around 10 cm thick (in several 2 cm steps over the summer season corresponding to the drainage depth; Vybornova 2017) is excavated yearly, with an estimated total of 30,000–38,000 m^3 peat extraction per year. The rewetting process was started in 2004,

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by blocking drainage ditches and building dams, in both cases using peat. The first part of the industrial peat mining ceased in 2008; other parts followed in 2009, 2011 and 2015. All excavation activities are to be ended before 2019, and 75 % of the former extracted area has been flooded stepwise since the start of the rewetting project (Czerwonka & Czerwonka 1985, Vanselow-Algan *et al.* 2015). The peatland is used as an experimental study site by the Institute of Soil Science at the University of Hamburg, and continuous measurements of GHG fluxes have been made since 2011 (Vanselow-Algan 2014, van Asperen 2015, Vybornova 2017).

This study was carried out on an area of approximately 10.2 ha that was flooded in 2009, hereafter called the 'rewetting area' (R). The study site is located in the north-western part of the former extraction area, and was flooded with rainwater in 2009. The rewetting area (Figure 1) consists of bare flooded areas (FL) and the surrounding bare peat dams (PD). More than 96 % of the total FL area is constantly flooded and soils are water-saturated over their whole depth. Indeed, only a few parts of the rewetting area become dry at the soil surface during long dry periods, which mostly occur in August. The peat which forms the dry wall of the PD area was strewn and compressed (50 cm high and 4 m wide) around the borders of the FL area in 2009 (Figure 1), and the drainage ditches were covered with peat material to reduce lateral water losses from the study site. This resulted in several bare dry peat dams with a total area of approximately 2.1 ha and a height difference between the dams and the flooded area of 50 cm. Today (in 2018) a secondary vegetation succession is occurring; vascular plants and birch began to colonise the rewetting area in 2012, and by 2016 occupied 1.7 % of the PD area and less than 1 % of the FL area.

In order to investigate the effects of water table level on GHG fluxes at Himmelmoor, flux measurements from the dry PD area were compared to flux measurements from the FL area (rewetted in 2009), as well as to flux measurements from an extraction (E) area located to the north of the rewetting area (R). Two main field campaigns have focused on the quantification and comparison of GHG fluxes from these different types of areas (PD, FL, E). In the first field campaign (August to November 2012), a Fourier Transform Infrared Spectrometer (FTIR analyser; Spectronus, Ecotech) was combined with two automated flux chambers which measured CO_2, N_2O and CH_4 fluxes at 60-minute intervals. During the 2012 field campaign, the different site types were not measured simultaneously; PD was measured in August and

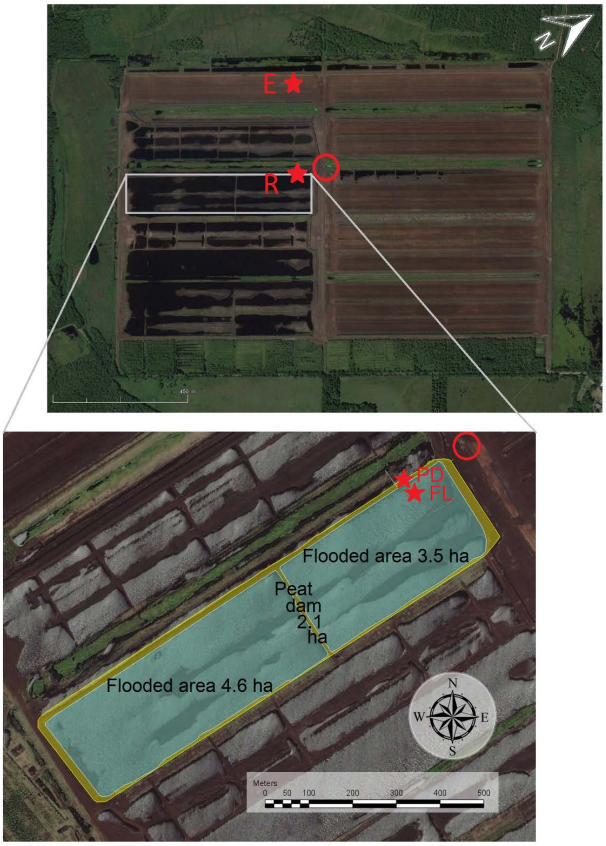

Figure 1. Aerial photograph of the Himmelmoor showing the extraction area (E) and the rewetting area (R) with its two study sites. Blue areas: flooded in 2009 (FL); grey area: peat dams (PD) around the borders of FL. Red stars indicate plot locations; red circle indicates the meteorological station at the centre of the bog. Modified from Google Earth (2014).

September, and FL was measured in October and November. The second campaign started in January 2014 and ended in May 2016, and consisted of weekly to biweekly flux chamber measurements of CO_2, N_2O and CH_4. During this second campaign a bare abandoned (since 2014) extraction (E) site 300 m north of the rewetting area was examined in addition to the two study areas (FL, PD) inside the rewetting area. The bare extraction site E had an area of 31 ha, and the upper soil layer was drained to a depth of 33 cm.

Instrumentation and environmental measurements

In order to monitor different environmental variables, a meteorological station was installed in the centre of the peatland in 2011. At 6.5 m height above the soil surface, the station measured the following variables: air temperature (Tair; model HMP45, Campbell

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Scientific Inc., USA), wind speed, wind direction and precipitation (Ws; model 05103-5, R.M. Young Company, USA). During the second field campaign in 2014–2016, the following additional variables were measured (bi)weekly. Soil temperature and soil redox potential (Ts, Eh) were measured at each PD and FL collar with soil sensors located at four different depths (2, 5, 10, 20 cm) using platinum electrodes (Hypnos III, MVH Consult, Netherlands). Redox potential was calculated by adding the electric potential from the calomel reference electrode to the potential measured by the platinum electrodes. A perforated PVC pipe (5 cm wide, 2 m long) was inserted vertically into the peat at each study site to measure the groundwater table level and water temperature inside the pipe at 30 min intervals using Mini-Diver sensors (Schlumberger Water Services, Netherlands). Furthermore, mixed soil samples from the first four horizons in three replicates were taken at each study site in May 2014 and were analysed for pH, total C and N, microbial C and N, and water content.

Continuous flux chamber measurements of CO_2, CH_4 and N_2O in 2012

The first set of flux chamber measurements was performed in the summer and autumn of 2012 using an FTIR analyser (Spectronus, Ecotech). During this campaign, no measurements were performed at the abandoned extraction site (E). Measurements were performed on the dry dam (PD) from 04 August to 20 September 2012, and on the flooded area (FL) from 04 October to 15 November 2012. In each case, two identical flux chambers (Chambers A and B; both closing once an hour) were deployed close to one another (< 5 m apart). The flux chambers (50 cm × 50 cm × 50 cm) were manufactured by KIT (Karlsruhe Institute of Technology, Germany) and each consisted of a stainless steel frame, UV-transparent acrylic top and sides (Acryl Glass XT solar, 3mm) and a vent tube. Transparency of the acrylic material was > 90 % in the UV and visible wavelength band (280–700 nm). The flux chambers were secured with clamps and rubber air strips to soil collars (10 cm deep, inserted one week before the start of the experiment). Two fans per flux chamber were running continuously to ensure well-mixed headspace air. An external manifold box was designed to automate chamber closure by a pneumatic system (once per hour per chamber), and with this the FTIR analyser could be connected to the two flux chambers in a closed loop. Air flow from the flux chambers to the FTIR analyser was initiated by a membrane pump placed behind the measurement cell, set to 1 L min^{-1}. Air flow was measured every 2 minutes continuously for 14 minutes in flow mode. Chamber opening and closure took place 2 and 14 minutes after start of the air flow, respectively. Sampling lines from the chambers were made of PTFE material and were tested for leaks regularly. Before being measured, air samples were dried by a Nafion dryer and by a column of magnesium perchlorate. Measurements were corrected for pressure and temperature variations as well as for cross-sensitivities (Hammer *et al.* 2013). Background measurements and a calibration routine were performed weekly or biweekly.

Manual opaque chamber measurements of CH_4, CO_2 and N_2O in 2014–2016

During the second campaign (January 2014 to May 2016), gas flux measurements were conducted at weekly or biweekly intervals using the manual static closed chamber technique. During chamber placement, the headspace of the chamber was completely isolated from the atmosphere. In order to avoid compression of the peat during the flux measurement, wooden boardwalks were constructed at the study areas one month before the measurements began. In contrast to the first campaign, all study sites (E, PD and FL) were measured on the same days. At all study sites, the GHG emissions were measured in three replicates. Three stainless steel collars (60 cm × 60 cm × 30 cm) with holes in their walls (to allow lateral water flow) were permanently inserted into the peat within the PD and E study sites, with approximately 30 cm distance between the collars. The box-shaped gas chambers were made of aluminium and had a volume of 0.11 m^3. The air inside the chamber headspace was mixed by a battery-operated fan, and the air temperature within the headspace volume was monitored. Before the measurement, two vent holes (diameter 4 cm) at the front site of each chamber were set open to prevent initial pressure shocks while setting the chamber on the collar, but were closed immediately afterwards (Schneider *et al.* 2009). At the FL site, three floating chambers (60 cm × 60 cm × 33 cm) were constructed near the boardwalk. At the PD and E sites, the chamber was placed in the water-filled frame on top of the collar, and after closing the two vent holes, six gas samples were taken from the headspace volume (0.11 m^3) after 0, 6, 12, 18, 24 and 30 minutes using 60 mL plastic syringes with a three-way stopcock. Gas sampling was done between 10:00 and 13:00 hrs., as recommended by different authors to prevent the diurnal flux variability affecting the measurement time series (Parkin & Venterea, 2010, Vanselow-Algan *et al.* 2015). The gas fluxes were measured three or four times per month during the

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growing season and biweekly during the winter season. In winter, as snow depth was less than 30 cm, the snow gas gradient method (Alm *et al.* 2007) was not required and GHG fluxes were measured in the headspace above the snow cover.

The gas samples taken from the chamber headspace were analysed for CO_2, N_2O and CH_4 as soon as possible (within two days after sampling) at the Institute of Soil Science (University of Hamburg) using a gas chromatograph (GC; Agilent 7890A, Agilent Technologies, USA) equipped with a flame ionisation detector (FID) and an electron capture detector (ECD). The samples were injected into the carrier gas stream via the septum module and transported to the analytical column, where they were separated into three analysed gas components depending on their retention times, and then measured by FID (CH_4, CO_2) and ECD (N_2O), respectively. Six standard gases (three mixed gases with the following CH_4 and CO_2 concentrations:1.79 ppm and 387.8 ppm, 209.7 ppm and 1005 ppm, 1004 ppm and 9710 ppm; and three N_2O mixtures: 0.29 ppm, 1.05 ppm and 1.55 ppm) were used for calibration by injecting them three times before and after the sample measurements of one field day. Two additional measurement tests were conducted at the beginning and at the end of the campaign to estimate the losses of gas concentration from the sampling syringes. In these, five plastic syringes were filled with different standard gases and measured daily for a one-week period. The tests suggested that an average of 2.13–9.1 % of the initial gas concentration in the syringes was lost per day in storage. Information on this loss was accounted for in the calculation of the gas concentrations.

Flux calculation

For both field campaigns, the CO_2, CH_4 and N_2O fluxes (F_{gas} in mol m^{-2} s^{-1}) were derived from the concentration increase in the chamber headspace during the chamber closure time by determining the slope parameter (the change in gas concentration over time; ∂_c/∂_t in mol mol^{-1} s^{-1}) using the ideal gas law as follows (Livingston & Hutchinson 1995):

$$F_{gas} = \frac{P_a \cdot V_{ch}}{R \cdot T \cdot A} \cdot \left(\frac{\partial_c}{\partial_t}\right) \quad [1]$$

where P_a is the air pressure during sampling (Pa), V_{ch} is the volume of the chamber headspace (m^3), R is the universal gas constant (8,314 Pa m^3 mol^{-1} K^{-1}), T is the headspace temperature at the beginning of sampling (K) and A is the soil area covered (m^2). The chamber headspace was calculated as the sum of chamber volume with total volume of the sampling lines and FTIR analyser measurement cell for the first campaign, and with the volume of each individual collar for the second campaign. The latter was calculated using the collar area and the mean distance between the soil surface, moss or snow cover and the water-filled frame on top of the collar.

Annual GHG fluxes were estimated for the second campaign (2014–2016). Daily average fluxes were calculated by averaging the fluxes for each of the three replicate plots, then multiplied by the length of each season (in days) to derive a seasonal gas budget. Since gas fluxes were only measured during daytime, these estimates did not consider daily variation. To account for the daily variation in the PD fluxes in summer (June–September, representing the months with the largest fluxes and monthly soil temperature > 13 °C at 10 cm depth), a correction based on the daily changes in magnitude observed in 2012 was applied:

$$F_{corr} = F_{daily} \times \frac{F_{mean}}{F_{max}} \quad [2]$$

where F_{corr} is the daily gas flux after correction, F_{daily} is the daily mean measured from 10:00 to 12:00 hrs during the second campaign, F_{mean} is the daily mean flux measured during the first campaign (in 2012), and F_{max} is the maximum gas flux measured at 12:00 hrs during the first campaign. Since no diurnal measurements were made in winter, no correction could be applied for this period. However, a previous campaign at Himmelmoor showed that wintertime CO_2, CH_4 and N_2O fluxes varied little over the day, on which basis it is expected that a correction is non-essential (Vanselow-Algan 2014). The annual CH_4 and N_2O fluxes were converted into CO_2 equivalents (CO_2 eq) using global warming potentials (GWP; over a 100-year time period with included climate-carbon feedbacks) of 34 and 298, respectively (Myhre *et al.* 2013), to estimate the total effect of the study area's emissions on the climate system.

Gas fluxes measured during the first campaign were derived from the concentration increase between 2 and 10 minutes after chamber closure by linear regression analysis. Flux measurements based on linear regression with R^2 values < 0.9 were not used. Flux standard deviations were derived from the propagated standard deviations of the regression slope.

The gas fluxes during the second campaign were determined by conducting a linear or non-linear regression on the time series using the MATLAB software (Matlab R2013a; Mathworks, USA) routine of Forbrich *et al.* (2010). The linear model was used to estimate the mean increase in concentration over the closure period, whereas the slope of exponential regression was used to determine the rate of initial

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concentration increase at the start of the measurement. In order to find the best fitting relationship between the gas concentration change and the regression, all datasets were checked using the MATLAB routine for errors and concentration anomalies, which were then removed from further calculations. Furthermore, the concentration detection limit was determined according to the gas chromatograph's precision, and only fluxes of chamber measurements with temperature changes of less than 5 °C were included in further analysis (Günther *et al.* 2014). More than 70 % of data passed these criteria, and the most frequent reason for rejecting a measurement was the suspected occurrence of ebullition (9 % of all gas fluxes from the FL site were thus excluded). As only six gas samples were analysed in this research to estimate the flux curve, the *Akaike* Information Criterion AIC*c* with small sample second order bias correction (Kutzbach *et al.* 2007) was used to determine whether the linear or non-linear model best represented the final slope for each measurement, as described in Vanselow-Algan (2014).

Statistical analysis

For the measurements of the second field campaign, the flux calculation was based on three replicates (three collars or three floating chambers) and expressed as a mean ± standard deviation for each study site before statistical analysis. Data were tested for a normal distribution using the Shapiro-Wilk test, and for homogeneity using the Levene's test. The distribution of GHG fluxes for all locations and the differences among the environmental data were tested using parametric and non-parametric one-way analysis of variance (ANOVA and Kruskal-Wallis test). In addition, the post-hoc comparison was conducted using Tukey's Honestly Significant Differences test, and correlation analyses (Spearman's rank-order and Pearson's correlation coefficients) were carried out to determine the effects of abiotic parameters on GHG fluxes. All calculations and statistics were computed using SPSS software (IBM SPSS Statistics 22, IBM, USA). Origin software was used for graphical analyses (Origin 9.1, OriginLab Corp, USA).

RESULTS

Environmental conditions

In the years of the study period (2012, 2014, 2015, 2016), the average annual mean air temperature and precipitation were 9.7 °C and 793 mm, respectively. Annual temperatures were consistently higher than the long-term average of 8.9 °C calculated for the 40-year reference period 1975–2016 (DWD, Quickborn meteorological station, 1 km from Himmelmoor). The maximum temperature during the study period (+ 35.9 °C) was measured in August 2012 and the minimum (-12.6 °C) occurred in January 2014. Compared to the long-term (1975–2016) mean annual rainfall of 781.4 mm, the conditions in 2014 were slightly drier than average, whereas 2015 saw conditions that were considerably wetter than average. The coldest and driest summer occurred in 2012 with a mean air temperature of 15.8 °C and 21 mm of precipitation (June–August) in comparison to the warm and wet conditions in the summers of 2014 and 2015 with 16.8 °C and 107 mm and 16.4 °C and 138 mm, respectively (Figure A1 in Appendix). Soil temperatures at 10 cm depth varied in 2014–2016 between -1 °C and +24 °C at the PD site, between +1 °C and +24 °C at the E site and between +2 °C and +26 °C at the FL site (Figure A2).

In general, flooding of the soil surface at the three studied FL subsites (for the period 2014–2016) was observed all year round except in August, and began after strong, prolonged rain events in autumn. Mean daily WTLs at all study sites were lower in 2014 than in subsequent measurement years (not measured before 2014). Water table level ranged from -102 to -66 cm relative to the surface of the PD site, from -4 cm to +54 cm relative to the surface of the FL site (the soil surface height in the PD site is 50 cm higher than that in the FL site), and from -71 to +1 cm relative to the surface of the E site. The WTL at the FL site ranged between +4.2 cm (September 2014) and +53 cm (February 2014). The mean annual WTL at the FL site was +20.8 cm in 2014 and +28.1 cm in 2015, showing drier conditions in 2014. In contrast, the WTL position at the PD and E sites was consistently below the surface, with a mean of -76 cm at the PD site and a mean of -33 cm at the E site.

The soil redox potentials measured at the PD and FL sites showed pronounced temporal variability depending on the local WTL as well as on the precipitation rates, with an annual mean at 10 cm depth of 271 mV at the PD site in comparison to an annual mean of -328 mV at the FL site. In general, smaller fluctuations in redox potential as well as fewer precipitation-induced alterations were observed for the PD site, with values ranging between +164 and +416 mV.

Soil bulk density was relatively low due to the high organic matter content, and was in agreement with other values for peat soils given in the literature (Périé & Ouimet 2008, Oleszczuk & Truba 2013, Hossain *et al.* 2015). The highest mean bulk density of 0.22 ± 0.01 g cm^{-3} was found in the upper soil layer

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of site PD, and was significantly higher than densities found at the extraction (0.14 ± 0.01 g cm^{-3}) and flooded (0.12 ± 0.01 g cm^{-3}) sites ($p = 0.014$), most probably as a result of soil compression by heavy peat extraction machinery. The dry peat dam site (adjacent to the FL area) was characterised by denser (45 % more dense) and more strongly decomposed (H8) peat compared to the E and the FL areas (H5–H7). The soil organic carbon stocks (C_{stocks}) estimated for the upper 100 cm of soil showed large variability (Table 1) and the main factor here was the soil bulk density and soil compaction in general.

Continuous flux chamber measurements using an FTIR analyser in 2012

During the first part of the field campaign, in August and September 2012, the flux chamber measurements were at the PD site. During October and November, the flux chamber measurements were at the FL site. During both periods, the chambers measured continuously (flux measurement once per hour per chamber). Monthly averaged diurnal flux measurements performed at the PD and FL sites are shown in Figure 2.

At the PD site, large CO_2 emissions of up to 22 µmol m^{-2} s^{-1} were observed, and a daily pattern with higher emissions during the day was found. CH_4 fluxes did not show a clear daily pattern, and showed mostly uptake in one chamber position and mostly emission in the other. N_2O emissions were very high (with daily mean fluxes of up to 23 nmol m^{-2} s^{-1}). The chamber positions differed largely in flux magnitude, especially for the gases CH_4 and N_2O, but both showed a decrease in emissions from August to September.

At the FL site, CO_2 emissions were very small for both positions and clearly decreased from October to November. Both FL chamber positions showed mostly CH_4 emissions and only little uptake, but no daily pattern was observed. N_2O emissions of up to 12 nmol m^{-2} s^{-1} were observed, but fluxes clearly decreased from October to November.

Manual opaque chamber measurements in 2014–2016

Carbon dioxide

During the second campaign (2014–2016), a strong seasonality in CO_2 emissions was observed at all sites, with significant differences ($p < 0.001$) between seasons and between the measurement years. Generally, the highest annual mean CO_2 flux and the strongest temporal variability between months were measured in 2014, while in 2015 the mean annual CO_2 flux was 31 % lower than in 2014 at all sites. A detailed overview of the measured CO_2 fluxes for each of the monitoring sites and collars is shown in Figure 3. All plots were found to be strong sources of CO_2 during the summer period, while wintertime emissions (December–February) were much lower, being close to zero at all monitoring plots, and even showing uptake at times.

For the PD site, emissions were higher in autumn than during spring for both years (2014 and 2015). In summer, the CO_2 emissions decreased during dry periods and, conversely, CO_2 emission peaks followed large rainfall events. Estimated daily CO_2 fluxes for the PD site ranged between 0.03 and 6.9 µmol m^{-2} s^{-1} over the study period, with an average annual mean of 1.8 µmol CO_2 m^{-2} s^{-1}.

At sites FL and E, the largest CO_2 emissions were observed in late summer and early autumn 2014, with average flux rates of 3.0 and 1.4 µmol m^{-2} s^{-1}, respectively, and the majority of fluxes ranging around zero. Analysis of the data indicated that a linear regression model provided a better fit to the calculated CO_2 fluxes in the majority (70 %) of cases when compared to an exponential regression model.

Methane

CH_4 fluxes measured over the 2.5-year study period showed strong temporal variability with significant ($p = 0.001$) seasonal and inter-annual differences. The mean CH_4 fluxes observed in 2014 at all sites were significantly different from the emissions observed in 2015 and 2016 ($p < 0.001$ and $p < 0.005$, respectively), but there was no significant difference in CH_4 emissions between 2015 and 2016 for all sites (Figure 4). Winter emissions in 2014 were significantly larger than those in 2015 and both were significantly smaller than summer fluxes for all plots ($p = 0.015$). Summer CH_4 emissions demonstrated a clear increase during wet periods, and emission peaks followed after strong precipitation events. The highest values of CH_4 emission were detected at the FL site, ranging between -12.8 and 46.3 nmol m^{-2} s^{-1} with a mean flux of 9.5 nmol m^{-2} s^{-1}. The PD and E plots exhibited generally low CH_4 fluxes with some irregular and sporadic outbursts. For example, although the estimated mean CH_4 emissions for the PD site were very low (2.7 nmol m^{-2} s^{-1}), clear methane flux peaks of up to 23.5 nmol m^{-2} s^{-1} were observed during the summer and autumn period. At site E, the fluxes ranged between -11.5 and 135.5 nmol m^{-2} s^{-1} (mean flux 5.8 nmol m^{-2} s^{-1}) and the largest CH_4 emissions were detected during spring and summer 2014, in addition to the high flux peaks following collar installation in February 2014 when irregular high fluxes of up to 193.5 nmol m^{-2} s^{-1} were measured.

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Table 1. Bulk density (ρ_b), pH, carbon density (ρ_c), SOC stocks, nitrogen density (N_{total}), N stocks, microbial C (C_{mic}), microbial N (N_{mic}) and C/N determined for all peat horizons and as a total mean for the upper 100 cm depth for each investigated soil profile, as sampled in May 2014 (for C/N additionally in May 2015 and 2016). Results are expressed as mean values ± standard deviation (n = 3).

Depth	ρ_b	pH_{CaCl2}	ρ_c	SOC_{stocks}	N_{total}	N_{stocks}	C_{mic}	N_{mic}	C/N		
(cm)	(g cm^{-3})	(-)	(kg m^{-3})	(t ha^{-1})	(kg m^{-3})	(t ha^{-1})	(µg cm^{-3})	(µg cm^{-3})	2014	2015	2016
FL: Flooded in 2009											
0–14	0.12±0.01	3.0	70.7±0.1	99.0±0.1	1.6±0.01	2.2±0.0	373.8±16.2	3.1±0.0	49:1	n.d.	46:1
14–32	n.d.	3.2	70.1±0.4*	126.2±0.7*	1.5±0.01*	2.8±0.0*	583.9±209.6*	1.1±0.0*	47:1	n.d.	46:1
32–44	n.d.	3.4	68.6±0.3*	82.4±0.3*	2.0±0.01*	2.3±0.0*	991.1±133.1*	0.0±0.0*	33:1	n.d.	35:1
44+	n.d.	3.6	65.1±0.3*	364.5±1.8*	2.6±0.01*	14.7±0.1*	827.0±50.9*	0.0±0.0*	31:1	n.d.	28:1
Total amount for the upper 1 m of soil:				**671.9±2.9**		**22.0±0.2**					
PD: Dry bare peat dam											
0–8	0.22±0.01	2.7	127.1±0.1	101.7±0.1	2.8±0.01	2.2±0.0	356.7±2.3	7.6±1.0	46:1	n.d.	46:1
8–28	0.18±0.02	2.7	105.7±0.1	211.4±0.0	2.4±0.00	4.9±0.0	339.6±2.5	2.5±0.1	36:1	n.d.	43:1
28–70	0.15±0.01	3.2	82.8±0.3	347.6±1.0	3.0±0.02	12.8±0.1	743.5±8.1	17.6±5.1	25:1	n.d.	27:1
70+	0.14±0.01	3.1	74.2±0.3	222.5±0.9	3.0±0.09	9.1±0.3	1295.1±13.2	20.4±2.0	23:1	n.d.	24:1
Total amount for the upper 1 m of soil:				**883.2±3.0**		**29.0±0.3**					
E: Bare extraction area											
0–10	0.14±0.00	3.1	74.2±0.7	74.2±0.7	1.82±0.03	1.8±0.0	368.8±7.1	9.3±1.0	42:1	42:1	42:1
10–42	0.10±0.01	3.4	60.6±0.1	193.8±0.1	1.35±0.01	4.3±0.0	341.7±16.3	0.0±0.0	39:1	40:1	45:1
42–82	0.11±0.01	3.4	63.8±0.1	255.4±0.1	1.26±0.01	5.1±0.1	699.3±18.9	0.0±0.0	37:1	41:1	50:1
82+	0.09±0.02	3.4	55.9±0.1	100.5±0.0	2.00±0.01	3.6±0.0	549.1±5.09	25.2±3.0	34:1	30:1	28:1
Total amount for the upper 1 m of soil:				**623.9±2.9**		**14.8±0.1**					

* bulk density determined for the first horizon (0–14 cm) was used for calculations.

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Nitrous oxide

Over the 2.5-year measurement period, N_2O fluxes displayed strong seasonal and inter-annual variability ($p < 0.001$ and $p < 0.001$, respectively). The highest N_2O emissions and the largest variance were seen in 2015, when fluxes of up to 13 nmol m^{-2} s^{-1} were observed. Analysis showed that N_2O fluxes in summer were similar between measurement years, whereas significant differences were observed in winter. This observation was made for individual PD and E plots but not for the FL site, where no significant seasonal differences in winter mean fluxes were found. N_2O fluxes ranged between 0.1 and 13.0 nmol m^{-2} s^{-1}, between -0.6 and 2.3 nmol m^{-2} s^{-1} and between -0.4 and 1.6 nmol m^{-2} s^{-1} at the PD, E and FL sites, respectively. As can be seen in Figure 5, during the late summer and autumn all monitoring sites were N_2O sources. Moreover, the summer N_2O emissions

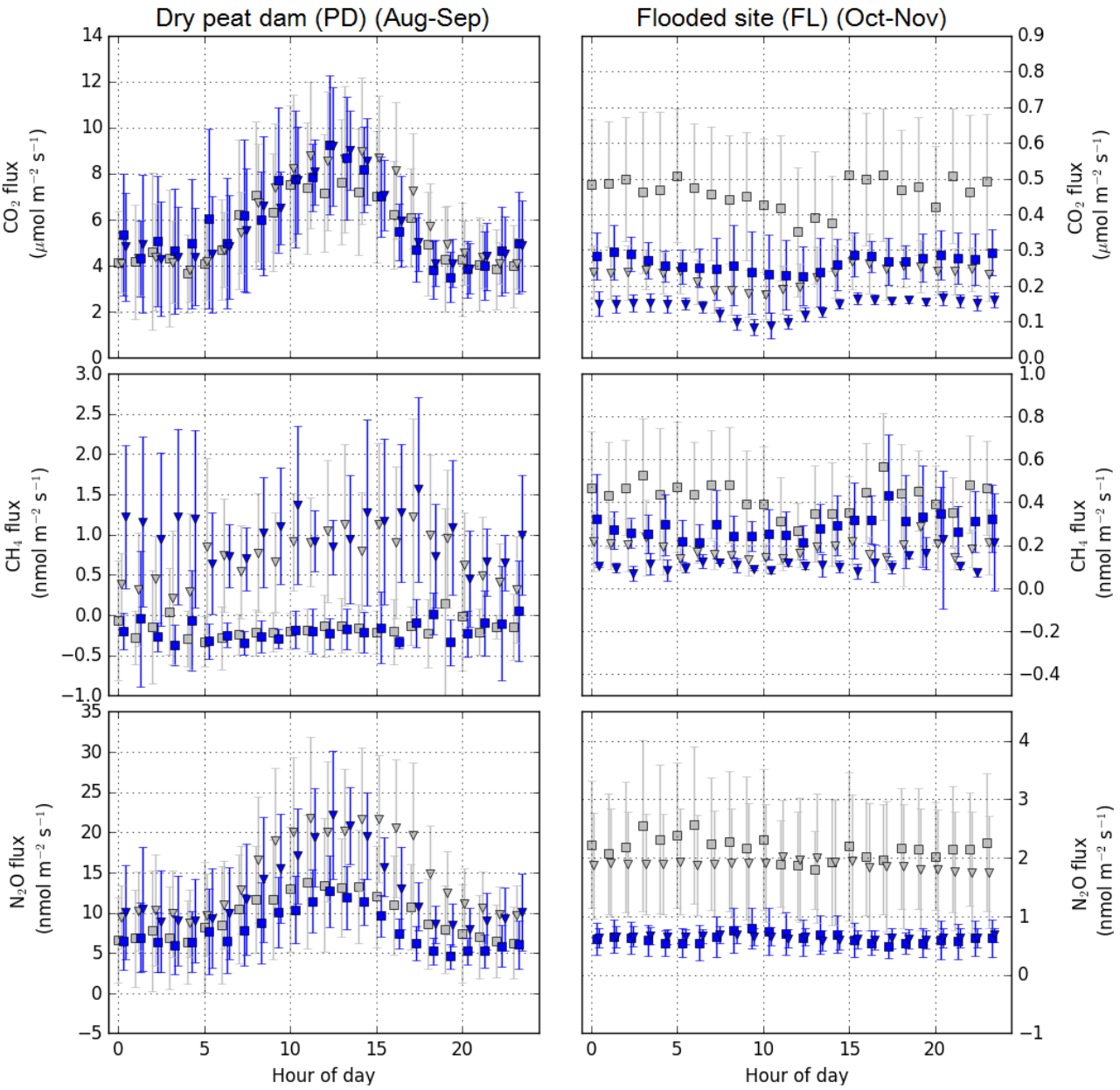

Figure 2. Daily averaged fluxes per month, location and gas. Left-hand plots show measurements from the dry peat dam (PD), measured in August and September 2012; right-hand plots show measurements from the flooded area (FL), measured in October and November 2012. Grey colour is for August (left) and October (right), blue colour is for September (left) and November (right). Different markers indicate different (fixed) locations. For these plots, the 5–95 percentile range of each monthly dataset was used. Please note the deviating y-scales. Positive flux numbers indicate emissions, negative flux numbers indicate uptake.

showed a clear increasing trend during dry periods. Emission peaks followed large rainfall events and thaws, as for the CO_2 fluxes. Statistical analysis revealed no significant difference between summer and autumn flux rates, although it did highlight the strong variability between summer, winter and spring emissions ($p < 0.001$).

Impact of abiotic conditions on GHG fluxes

For the flux measurements from the second campaign, the impact of environmental conditions on GHG fluxes was studied. Fluxes of CO_2, CH_4 and N_2O were found to display clear relationships with soil and air temperature, with the strongest positive correlations being observed at the PD site between soil temperature and CO_2 ($r = 0.7$) and N_2O ($r = 0.75$) fluxes. Heavy rainfall events were also found to have some impact on the CO_2 and N_2O fluxes at all study plots ($p < 0.001$), with larger emissions following periods of heavy precipitation.

In contrast to the above, a strong negative relationship between GHG fluxes and WTL was observed and, in general, this relationship was strongest for CO_2 ($r = -0.6$) followed by N_2O ($r = -0.55$) and lowest for CH_4 ($r = -0.4$). Additionally, WTL was determined to be an important control factor on CO_2 and N_2O emissions for the PD plots ($r = -0.5$ for both gases). The soil redox potential displayed a weak relationship with GHG fluxes for the whole study area, whereas a negative relationship was found for CH_4 fluxes ($r = -0.5$).

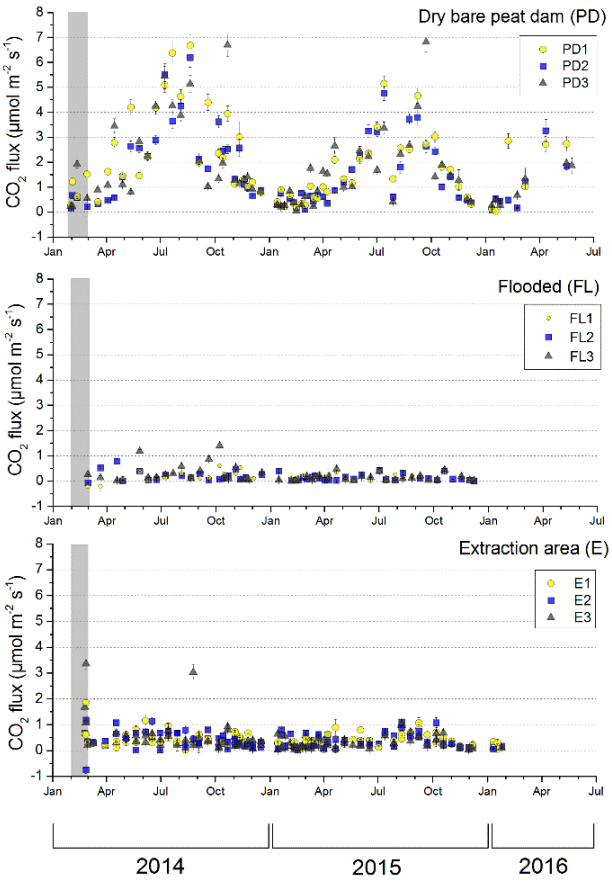

Figure 3. CO_2 fluxes (µmol m^{-2} s^{-1}) measured using closed chambers at the dry peat dam (PD), the flooded (FL) and extraction (E) positions over the second campaign 2014–2016. Positive values indicate a loss of CO_2 to the atmosphere and negative values indicate CO_2 uptake by the soil. Each replicate plot is shown separately. The grey area indicates the period of possible disturbance effects by inserting the collars. Error bars indicate standard errors of fluxes.

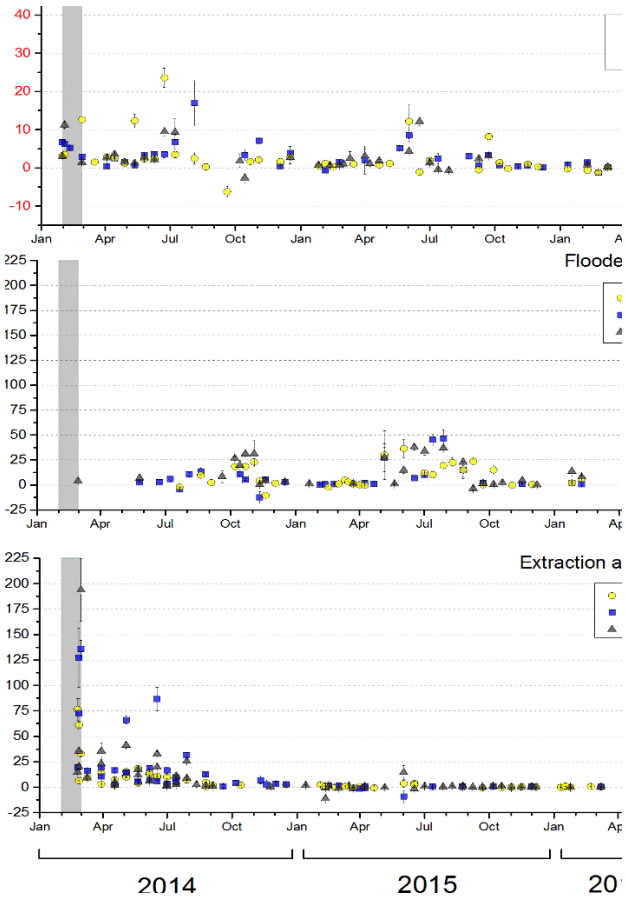

Figure 4. CH_4 fluxes (nmol m^{-2} s^{-1}) measured using closed chambers at the peat dam (PD), flooded (FL) and extraction (E) positions over the second campaign. Positive values indicate a loss of CH_4 to the atmosphere, and negative values indicate uptake by the soil. Each replicate plot is shown separately. Note that the PD site has narrower Y-Axis limits (in red). The grey area shows the period of possible disturbance effects after inserting the collars. Error bars indicate standard errors of fluxes.

Correlation of the CO_2 and N_2O fluxes

Overall, a significant correlation between CO_2 and N_2O emissions was identified for all sites ($p = 0.001$). For the years 2014–2016, the strongest correlation ($r = 0.9$) between these fluxes was observed at the dry peat dam site (PD). Furthermore, the observed degree of correlation in these data was very similar to that observed in 2012. As shown in Figure 6, the CO_2:N_2O emission ratio varied from year to year; in 2012 the ratio (473:1) was lower than in 2014 (717:1) and 2015 (722:1). A similar tendency was determined for the flooded site (FL), where the CO_2:N_2O ratio increased from 200:1 in 2012 to 704:1 in 2015. In contrast, at the extraction site (E; no measurements performed in 2012), the CO_2:N_2O ratio remained constant at 1440:1 over the 2014–2016 period.

A good correlation between CO_2 and N_2O fluxes, such as that observed in the PD plot, indicates gas sources that respond similarly to environmental variables such as soil temperature, WTL, soil redox potential etc. Analysis found that the correlation between CO_2 and CH_4 fluxes was significant only at the flooded (FL) plots, and no correlation between CO_2 and CH_4 fluxes was found for the PD site: this is likely to be due to the irregular emissions and low CH_4 fluxes observed at these plots.

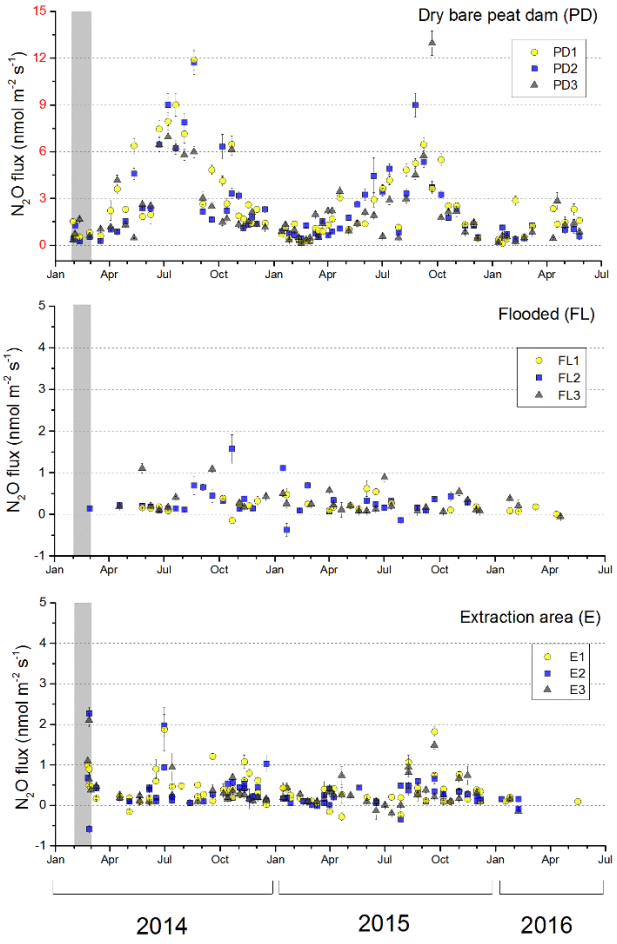

Figure 5. N_2O fluxes (nmol m^{-2} s^{-1}) measured using closed chambers at peat dam (PD), flooded (FL) and extraction (E) positions over the second campaign. Positive values indicate a loss of N_2O to the atmosphere, and negative values indicate N_2O uptake by the soil. Each replicate plot is shown separately. Note that plot PD has broader Y-Axis limits (in red). The grey area shows the period of possible disturbance effects after inserting the collars. Error bars indicate standard errors of fluxes.

Annual GHG emissions

In general, all study sites showed positive annual GHG emissions. However, the lowest annual GHG emission was identified at the FL area. For a 24-month period (2014–2015), the emissions including all three gases were considerably higher for the PD plot in comparison to the E and FL plots (Table 2). The mean annual GHG emission (in CO_2-eq) over the study area, including the CO_2, N_2O and CH_4 emissions, ranged between 4.7 ± 1.8 t ha^{-1} $year^{-1}$ (FL plot) and 31.5 ± 6.1 t ha^{-1} $year^{-1}$ (PD plot) for the R area and amounted to 7.4 ± 2.7 t ha^{-1} $year^{-1}$ for the E area. With respect to the total rewetting area in Himmelmoor, the emissions of the FL area, extrapolated to the total number of hectares, are considerably lower (in %) than the emissions of the area-extrapolated dry peat dam (PD). On the whole, annual GHG emissions were driven mostly by the CO_2 exchange (50–69 %), and the contribution of N_2O and CH_4 ranged from 3 % to 33 % and from 1 % to 30 %, respectively.

DISCUSSION

The special rewetting management at the Himmelmoor bog, i.e. ditch blocking and creation of polders surrounded by peat dams, has resulted in a more stable water level above the soil surface, less temporal variation in soil temperature, lower soil pH, lower humidification grades and a reduced bulk density at the peat surface of the FL site in comparison to the E site. This change is consistent with the findings of Couwenberg (2011), and supports the idea that the increase of supporting pore water pressure after flooding prevents further peat compaction and subsidence. Nevertheless, opposite effects were observed in the accompanying peat dams where heavy machinery was used to create stable impervious dams. A number of studies have suggested that peat subsidence and loss of carbon are

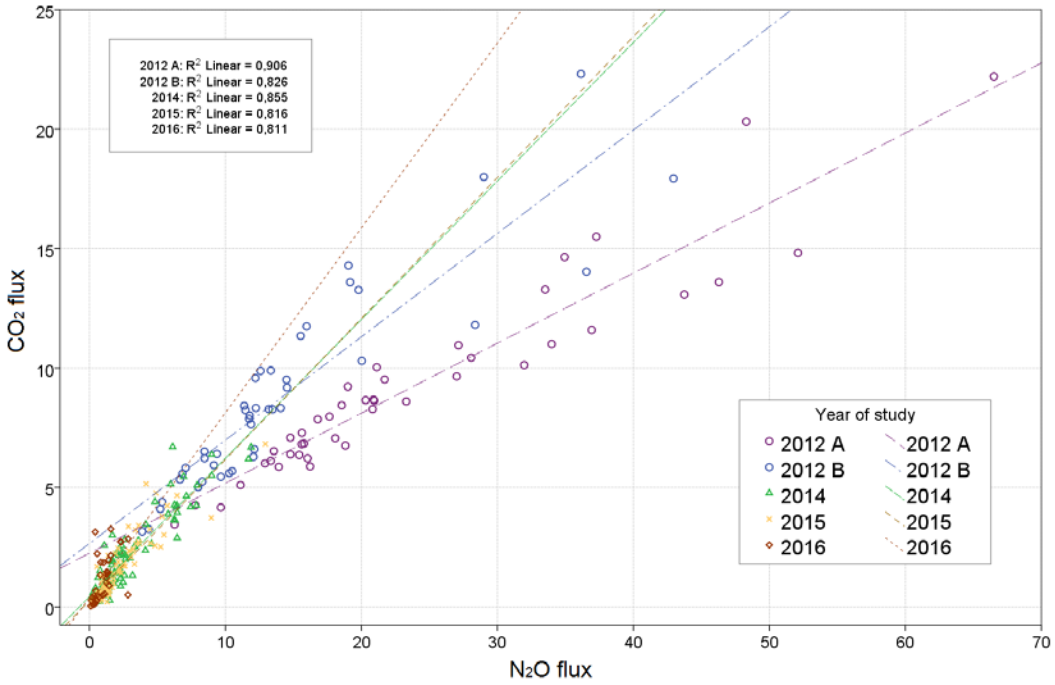

Figure 6. The correlation between CO_2 (µmol m^{-2} s^{-1}) and N_2O (nmol m^{-2} s^{-1}) fluxes using closed metal chambers (triangles, crosses and diamonds; second field campaign) and a FTIR flux chamber setup (open circles; first field campaign) for the PD site in the years 2012, 2014, 2015 and 2016. The measurements from 2014, 2015 and 2016 represent daytime measurements, while the measurements from 2012 also include night-time measurements.

the main consequences of peat oxidisation and drainage (Ketcheson & Price 2011, Hooijer *et al.* 2012). For this reason, the majority of investigations have focused on the positive impacts of peat dams, i.e., the rapid re-establishment of high water levels and past hydrological conditions (Jaenicke *et al.* 2010, Ketcheson & Price 2011, Gonzalez *et al.* 2013, Ritzema *et al.* 2014). The bulk density and resulting carbon and nitrogen stocks measured at the FL site were lower than in the PD site, but still up to ten times higher compared to other rewetted mires described in the literature (Chambers *et al.* 2011, Harpenslager *et al.* 2015, Gaudig *et al.* 2017). More than 25 years after closing the ditches in the Himmelmoor and 5 years after flooding, the evaluated soil conditions still remain affected by the impact of former management practices. According to different authors (Bain *et al.* 2011, Samaritani *et al.* 2011, Schimelpfenig *et al.* 2014), this outcome can be expected, and the full recovery of soil conditions after ditch refilling can take decades to centuries to occur.

CO_2 fluxes

CO_2 fluxes measured at a dry peat dam (PD) during the period 2014–2016 showed a strong seasonal cycle with highest emissions in the late summer months (July, August, September) and lowest emissions in the winter months (January, February). The seasonal trend is expected to be temperature-related: soil temperatures at 10 cm depth showed a high correlation to PD CO_2 fluxes. Annually, the averaged measured fluxes ranged from 0.03 to 6.9 µmol m^{-2} s^{-1}. The measurements in August and September 2012 showed similar emission magnitudes as measured in the summers of 2014 and 2015. These measurements also revealed a strong diurnal cycle with daytime fluxes double those during night-time. These diurnal flux measurements were used to correct the summer emission flux estimates for PD areas.

CO_2 fluxes measured at the flooded area (FL) during 2014–2016 were consistently lower than measured at the PD sites, especially in summer when emissions were sometimes ten times smaller. The measurements showed no seasonal cycle and ranged between -0.5 and 1.4 µmol m^{-2} s^{-1} (mean 0.15 µmol m^{-2} s^{-1}). In 2012, as well as in 2014–2016, strong spatial variability was observed with emissions measured a few metres apart sometimes differing by a factor of three. Fluxes measured in October and November 2012 were similar to those measured in the autumns of 2014 and 2015, and also revealed that no significant diurnal cycle was present.

The CO_2 fluxes measured at the abandoned extraction site (E) were in general larger than those

Table 2. Mean GHG emissions for the years 2014 and 2015 for both locations inside the rewetting area (FL, PD) and the extraction area (E), converted into CO_2 equivalents according to the global warming potentials (GWP for a 100-year period). PD means (red type) are corrected values and were estimated using Equation 2. The values given are annual means calculated from three replicates ± standard deviation.

Subsite	Year	**FCO_2-eq** (t ha^{-1} $year^{-1}$)			**Sum**	Area	Area-weighted average annual emission as CO_2 equivalent
		CO_2	**CH_4**	**N_2O**		(%)	(t ha^{-1} $year^{-1}$)
Rewetting area (R)						100	***10.4±2.6***
FL: area flooded in 2009	2014 2015	2.86±1.12 1.85±0.50	0.95±0.68 1.50±0.53	1.25±0.48 1.06±0.32	5.07±2.29 4.41±1.35	79	*3.8±1.4*
PD: dry bare peat dam	2014 2015	24.39±4.15 20.22±3.99	0.69±0.24 0.41±0.16	9.99±1.77 7.29±2.00	35.07±6.16 27.93±6.15	21	*6.6±1.2*
Extraction area (E)						100	***7.4±2.7***
E: bare extraction area	2014 2015	6.03±1.64 4.69±0.96	1.07±0.59 0.11±0.15	1.39±0.45 1.08±0.34	8.80±2.55 5.86±1.75		*7.4±2.7*

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measured at the FL site, and ranged between -0.9 and 3.0 µmol m^{-2} s^{-1} (mean 0.35 µmol m^{-2} s^{-1}). No diurnal measurements were performed here to assess a possible diurnal cycle.

A complete comparison of the measured PD, FL, and E CO_2 fluxes in Himmelmoor with literature values is restricted because no previous studies have examined GHG fluxes from peat dams (PD). Nevertheless, comparing the observed peat dam (PD) emissions (0.03–6.9 µmol m^{-2} s^{-1}) with results reported from other peatland studies shows that they are substantially higher than published values for other ombrotrophic bogs (0.8–3.6 µmol m^{-2} s^{-1}), drained managed peatland areas (0.6–1.2 µmol m^{-2} s^{-1}), peat extraction areas (0.5–3.0 µmol m^{-2} s^{-1}) and tropical peatlands under extraction (3.4–6.1 µmol m^{-2} s^{-1}) (Byrne *et al.* 2004, IPCC 2006, Alm *et al.* 2007, Blodau *et al.* 2007, Couwenberg 2011). The CO_2 emissions observed at the flooded plot (FL) are within the range reported in literature for flooded cutaway bogs in (temperate) Europe with emissions of 466 g CO_2 m^{-2} a^{-1} (average 0.33 µmol m^{-2} s^{-1}) (Drösler *et al.* 2008, Wilson *et al.* 2007). The low CO_2 emissions observed at the flooded plot (FL) in this study are likely to have been caused by the anoxic conditions, which are unfavourable for decomposition processes (Wilson *et al.* 2007, Landry & Rochefort 2012, Wilson *et al.* 2016). The emissions measured at the abandoned excavation area (E) are similar to those reported for other abandoned (Järveoja *et al.* 2016) and active (Wilson *et al.* 2015) extraction areas. For example, Shurpali *et al.* (2008) reported that emissions at a cutover bare peat area in eastern Finland ranged from 0.3 to 0.71 µmol CO_2 m^{-2} s^{-1}, and that CO_2 emissions were three times lower here than in areas cultivated with reed canary grass. Nevertheless, Järveoja *et al.* (2016) observed that CO_2 fluxes from an abandoned bare peat extraction area (mean flux 0.83 µmol m^{-2} s^{-1}) were 50 % higher than from areas which had undergone three years of restoration with *Sphagnum* and *Eriophorum* vegetation.

Overall, the CO_2 flux measurements support the hypothesis that peat dams (PD) are hotspots of CO_2 emissions in comparison to the flooded (FL) and extraction (E) areas. Furthermore, considering the similar magnitude of fluxes between 2012, 2014, 2015 and 2016, a reduction in CO_2 emissions over time is not expected. Nevertheless, with ongoing vegetation development, photosynthetic CO_2 uptake might change this net CO_2 flux in the future.

CH_4 fluxes

Observed CH_4 fluxes from all areas were highly variable, and uptake as well as emission was observed. During both field campaigns, FL uptake and emissions were higher, with mean CH_4 fluxes of 9.5 nmol m^{-2} s^{-1}, while PD showed mean fluxes of 2.7 nmol m^{-2} s^{-1}. However, for the 2014–2016 measurements it is possible that, given the size of the chambers and the sampling period of 30 minutes at midday, low CH_4 fluxes were not always detected. Both campaigns indicated high spatial variability with uptake and emission taking place close to each other. No clear diurnal pattern could be observed in 2012 and, although the majority of the observed CH_4 uptake at flooded and drained sites occurred over the summer and autumn periods, no clear annual trend could be observed in the years 2014 and 2015 in either the PD or the FL sites.

Mean annual CH_4 emissions at the flooded site FL (9.5 nmol m^{-2} s^{-1}, max 49 nmol m^{-2} s^{-1}) are comparable to those reported for other rewetted peatlands, where CH_4 emission rates range from 4 to 125 nmol m^{-2} s^{-1} (Wilson *et al.* 2013, Cooper *et al.* 2014, Günther *et al.* 2015, Karki *et al.* 2016). However, the CH_4 fluxes observed in this study are at the lower end of the range, probably as a consequence of a high C:N ratio in the peat (Klemedtsson *et al.* 2005).

The results also indicate that CH_4 emissions increase after (the first five years of) rewetting. The CH_4 emissions were 22–53 % higher at the FL site than at the E site, and it is assumed that the abandoned E site acts like the pre-rewetting site. Furthermore, the fact that the majority of the observed CH_4 uptake at the FL and E sites occurred over the summer and autumn periods may point to biological soil uptake, which is usually larger during periods characterised by warmer temperatures (Pfeiffer 1998). The large spatial and temporal variability of CH_4 emissions from rewetted peatlands has been reported in other studies, being linked to factors such as site-specific water table level (Couwenberg 2011), time since rewetting (Tuittila *et al.* 2000, Waddington & Day 2007) and soil environmental conditions (Günther *et al.* 2015).

Overall, the CH_4 flux measurements indicated that peat dams (PD) emit more CH_4 than the flooded (FL) or abandoned (E) sites. Due to the high spatial and temporal variability, it is unclear if an increasing or decreasing trend is present, and thus whether the presented flux estimates might increase or decrease over the coming years. Nevertheless, an increase in CH_4 emissions over future decades can be expected due to the secondary plant succession and the slow re-establishment of *Eriophorum* plants with aerenchymatous tissue in the FL (Abdalla *et al.* 2016); on the other hand, a decrease is also possible due to the transport of oxygen into the root zone (Laanbroek 2010).

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N_2O fluxes

The 2014–2016 measurements showed high PD N_2O fluxes ranging from 0.1 to 13 nmol m^{-2} s^{-1}. The measurements showed a strong seasonal trend with highest fluxes measured in summer and lowest fluxes in winter, which is also well represented by the high correlation found between N_2O fluxes and soil temperatures ($r = 0.75$). Averaged PD N_2O flux measurements from August and September 2012 ranged up to 23 nmol m^{-2} s^{-1}, but peaks up to 66 nmol m^{-2} s^{-1} have been observed. These fluxes are clearly higher than the August and September PD fluxes measured in 2014 and 2015. The measurements on the PD from the summer of 2012 showed a clear diurnal pattern with 2–3 times higher fluxes during the day. The decreasing fluxes from August to September, part of the strong seasonal trend, show that the day versus night flux difference is likely to become smaller in winter.

Consistently lower N_2O emissions were measured at the FL site (as well as at the E site) than on the peat dam. Mean N_2O emissions from these two plots (0.24 nmol m^{-2} s^{-1} for FL, 0.26 nmol m^{-2} s^{-1} for E) were 3–10 times lower than observed at the dry PD area. The N_2O emissions measured over a 2.5-year period (2014–2016) showed no clear annual pattern at FL and E sites, and ranged between -0.4 and 1.6 nmol m^{-2} s^{-1} and -0.6 and 2.3 nmol m^{-2} s^{-1}, respectively. The measurements from the autumn of 2012 at the FL site also showed lower N_2O fluxes in comparison to the PD site, but the fluxes were higher than measured in the autumns of 2014 and 2015. In contrast to the PD area, no diurnal N_2O emission pattern was observed at the FL site.

It was also found that, although the N_2O fluxes from the E and FL sites were generally small, the abandoned extraction area E emitted 14 % more N_2O annually than the now flooded site FL five years after rewetting. This supports the suggestion that N_2O emissions may be of minor importance under anoxic conditions; however, it has to be considered that denitrification may result in higher N_2 release, and hence in higher other gaseous N losses under anaerobic conditions (Velty *et al.* 2007, Ussiri & Lal 2013).

Compared to the existing literature, the N_2O emissions observed at the PD area in 2012 (mean 22 nmol m^{-2} s^{-1}) are much higher than those reported for other drained bogs and peat extraction areas (Maljanen *et al.* 2010, Jordan *et al.* 2016). The reason for this difference is unclear, although it is possible that high rates of nitrification and the consequent high nitrate (NO_3^-) content of the peat samples (Aerts 1997, Brumme *et al.* 1999) as well as the dry conditions (Schiller & Hastie 1994) and low soil pH (Aerts 1997) may promote increased N_2O emissions, inhibiting both the amount of N_2O reductase and N_2 production. Wrage *et al.* (2001) hypothesise that the dry oxic conditions in peat dams favour nitrification processes because nitrate is the substrate for peak N_2O emissions produced by denitrification (Lee *et al.* 2017). This hypothesis is supported by the high nitrate concentrations found in peat from the PD site, as well as the C/N value of 25 in the middle and lower horizons which is very suitable for soil microorganisms (Maljanen *et al.* 2010).

N_2O fluxes measured at the flooded (FL) sites during both campaigns were similar to those given in the literature (Karki *et al.* 2016, Jordan *et al.* 2016). Both the PD site and the FL site revealed high spatial variability of N_2O fluxes, as has been reported previously for other peatland microtopes (Landry & Rochefort 2012). For example, the 2014–2016 measurements identified clear differences in N_2O fluxes from the peat dam site across replicate collars located only a few metres apart. In addition, chamber measurements clearly showed lower N_2O emissions from one collar at the FL site than from the other two collars. Differences between chamber positions in the FL area in October and November 2012 were even larger; chamber B consistently showed lower N_2O and CO_2 emissions, which points to consistently lower levels of biological activity at this specific location. This spatial heterogeneity in N_2O flux magnitude is an important aspect to consider when upscaling flux measurements.

Temporal change in CO_2 and N_2O fluxes

The occurrence of a temporal trend between the different measurement years was identified, especially for N_2O and CO_2 emissions. At the peat dam, CO_2 and N_2O fluxes both showed a clear reduction over the course of the 2014–2016 study period and were markedly lower than fluxes observed in the summer of 2012. Furthermore, a consistent decrease in the CO_2:N_2O emissions ratio over time was evident. While it is possible that spatial variability caused part of the difference in the magnitude of observed fluxes between 2012 and 2014, the most likely reason for the observed decreasing trend in emissions at the peat dam and the flooded site is that it reflects a shift in soil conditions at these sites. The bare peat dams were constructed in 2009, at which point conditions became favourable for mineralisation and oxic decomposition, resulting in high CO_2 and N_2O fluxes. Over the years, it is likely that the amount of fresh readily available carbon and nitrogen has decreased, resulting in a subsequent reduction of CO_2 and N_2O fluxes, and explaining the large difference in observed flux

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magnitudes over the years 2012, 2014, 2015 and 2016. Another possible explanation for the reduction in N_2O emissions over time could be the higher C:N ratio (37–46:1; Klemedtsson *et al.* 2005) which was measured in peat samples from the upper soil horizons. The C:N ratio in the studied peat samples from the topsoil is higher than 25:1, meaning that less nitrogen is available here and that the decomposition process in the upper 30 cm of peat is slower. The fact that the upper 28 cm of peat at the PD site is strongly humified might also support the above theory. The changing ratio of CO_2 to N_2O indicates that the time frame of the mineralisation process may be different for the two gases. Moreover, the high amount of CO_2 produced and emitted to the atmosphere during mineralisation acts as an important index of soil biological activity.

Annual GHG estimates: considerations and assumptions

The estimated annual GHG fluxes, based on the measurement years 2014 and 2015, showed that the PD plot is a considerably (85 %) greater source of GHG than the FL plot (Table 2). It is interesting that, even though FL covers almost five times the area of PD, the absolute GHG emissions of the PD area still contribute most of the total emissions from the R area. Nevertheless, in interpreting these estimates, the following important assumptions and aspects should be considered:

(1) Concerning the CO_2 fluxes it should be noted that, as a result of the use of the opaque chamber measurement technique (second campaign in 2014–2016) and because bare soil locations were measured (first campaign in 2012), soil respiration (only) was examined and assumed to be equal to the net ecosystem exchange (NEE) of CO_2. Therefore, the increasing presence of vegetation (from 0 % of the study area in 2012 to 3 % in 2018) as a consequence of natural succession as well as rewetting, and its influence on the GHG budget, is not accounted for.

(2) A second important point is that (corrected) daytime GHG fluxes were used for upscaling, thereby neglecting possible diurnal variation. While the observations of the first field campaign (Figure 2) showed that diurnal variation in GHG fluxes is almost absent for the FL site and negligible for the PD CH_4 emissions, they also showed that the PD N_2O and CO_2 fluxes varied significantly over the day in August and September. The observed diurnal variation in 2012 was used to correct the summer (June–September) daily flux estimates of the lower flux magnitudes at night. This correction reduced the PD emission estimate for these summer months by 32 % (for CO_2) and 36 % (for CH_4) and the annual PD budget by 15 and 17 % respectively, in comparison to uncorrected estimates (Figure 7). Since year-round diurnal flux measurements are not available, the extent of any diurnal pattern in the winter months remains unknown and, therefore, a similar correction has not been applied for these months. Hypothesising an unlikely similar variation over all months would reduce the annual CO_2 and CH_4 budgets (in CO_2-eq) for PD by 15 % and 16 % to 22.3 t ha^{-1} $year^{-1}$ and 0.6 t ha^{-1} $year^{-1}$, respectively (Table 2), but it would remain a significantly larger GHG source than the FL and E sites. However, smaller daily variation is to be expected in winter; a previous campaign in Himmelmoor showed that wintertime CO_2, CH_4 and N_2O fluxes do not vary greatly over the day (Vanselow-Algan 2014).

(3) A third important point is the strong spatial heterogeneity between the different chambers located on the same type of site (PD, FL, E), which was observed mostly for N_2O and CH_4, and which could bias the GHG budget presented.

Additional sources of uncertainty are:

(4) the non-regular sampling conducted during the second campaign; and

(5) the unusually large fluxes measured in February 2014, which were possibly caused by disturbance of the soil during collar installation and are likely to have resulted in over-estimation of the winter fluxes in 2014.

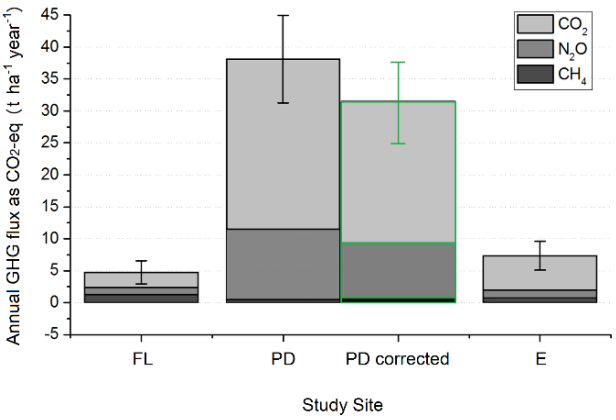

Figure 7. Estimated annual mean GHG balances expressed as CO_2-eq (t ha^{-1} $year^{-1}$) for the three study sites: flooded (FL), peat dam (PD), and extraction (E). Corrected values for PD (Equation 2) are shown (outlined in green) in the 'PD corrected' column. Positive flux numbers indicate emissions, negative flux numbers indicate uptake.

Implications for peatland rewetting practices

Despite the uncertainties in the GHG budget estimate, this study suggests that dry bare peat dams are hotspots of CO_2 and N_2O emissions, and strongly reduce the overall climate mitigation potential of rewetting actions during the first years of flooding. During future rewetting activities which involve the use of peat dams like those at Himmelmoor, the enhancing effect on GHG fluxes should be considered. Figure 8 shows the trade-off between the potential area-weighted mean GHG emissions (PAWM) and the relative proportions of FL and PD areas, and the implications for possible total GHG emissions during peatland rewetting, based on the results from Himmelmoor. A suggested cost effective action to mitigate the negative effects of peat dams is to reduce their height and width, and thus the area of dry peat (including the tops of the dams), in order to prevent erosion and oxidation of the peat material and reduce GHG emissions. Furthermore, the recolonisation of peat dams by mire or heath plant communities should be stimulated, for example by revegetation of bare peat with *Sphagnum* moss or heath.

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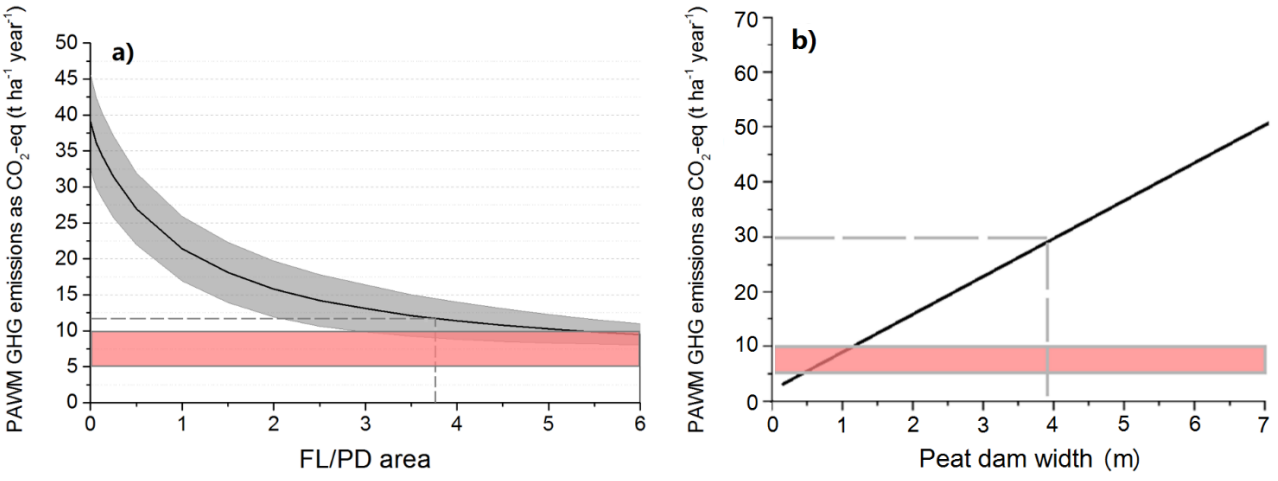

Figure 8. The relationship between estimated "potential area-weighted mean" (PAWM) GHG emissions and the possible distribution of peat dams in the R area: a) for different FL/PD area fractions and b) for different peat dam widths. Thin grey dashed lines indicate the current Himmelmoor FL/PD ratio (with a current peat dam width of 3.8 m) and the observed GHG emissions. Grey area in the left-hand graphic indicates standard deviation of mean GHG emissions. Red areas indicate calculated GHG emissions for the extraction (E) site for a better comparison with emissions of R area.

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Appendix

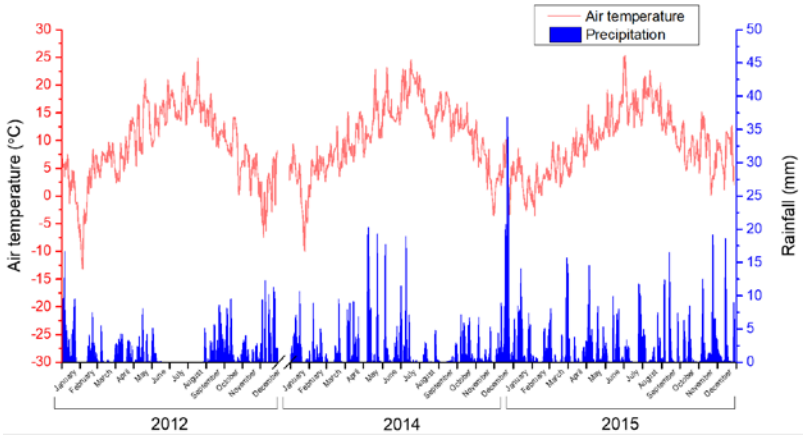

Figure A1. Daily means of air temperature (°C) and daily sums of rainfall (mm) in 2012, 2014 and 2015.

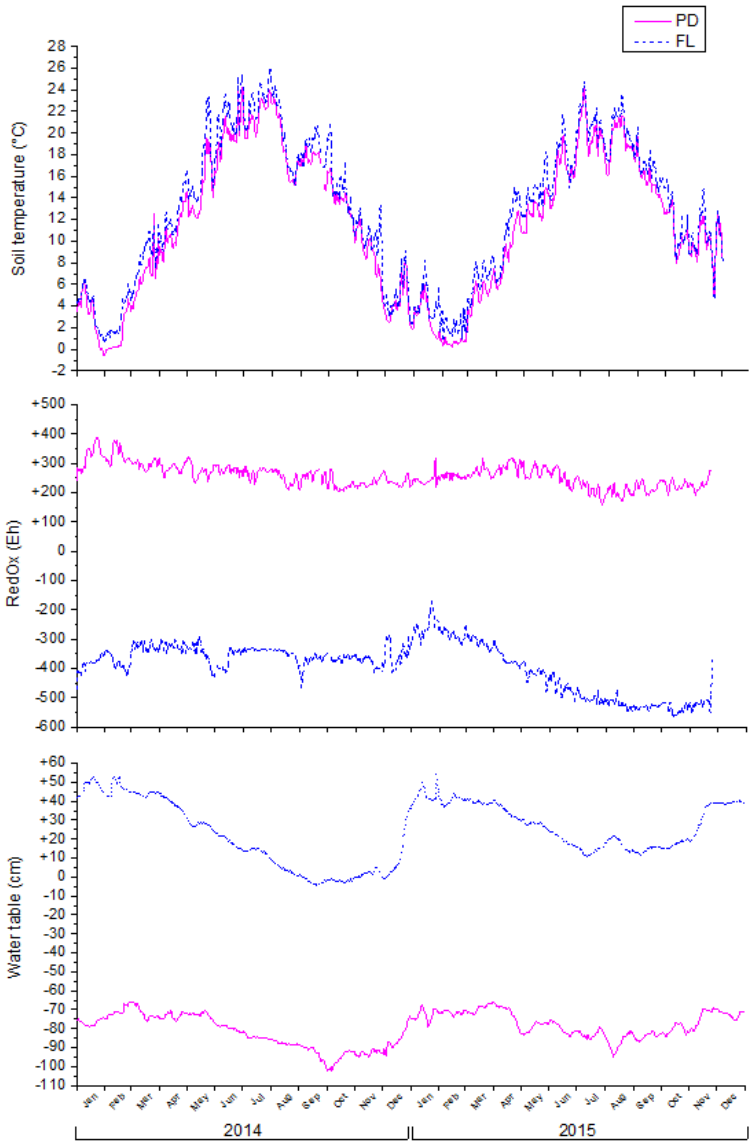

Figure A2. Daily means of soil temperature (°C), RedOx potential (Eh) and water table level (cm) in bare peat dam (PD, pink solid line) and flooded area (FL, blue dashed line) microsites in 2014 and 2015.

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