Wetlands Ecol Manage (2018) 26:87–102
https://doi.org/10.1007/s11273-017-9555-x

ORIGINAL PAPER

Changes of methane and nitrous oxide emissions in a transition bog in central Germany (German National Park Harz Mountains) after rewetting

Katja Osterloh · Nadine Tauchnitz · Oliver Spott · Johannes Hepp · Sabine Bernsdorf · Ralph Meissner

Received: 23 June 2016 / Accepted: 17 May 2017 / Published online: 23 May 2017

Abstract During the last decades, various renaturation programmes have been initialized to recover nutrient sink and ecological functions of peatlands by rewetting. Rewetting, however, often results in the formation of hotspots for methane (CH_4) emissions and in temporal dieback of local vegetation. The present study aimed at quantifying changes of CH_4 and nitrous oxide (N_2O) emissions in a peatland currently under continuous rewetting conditions. Emissions where studied at a permanently flooded site and a non-flooded peat site with fluctuating water tables by using common closed chamber method. The permanently flooded site revealed extremely high CH_4 emissions (up to 1195 mg C m^{-2} d^{-1}) which were positively correlated with temperature, nutrient content, dissolved organic carbon and nitrogen concentration of the peat soil water. In contrast, the non-flooded peat site, with lower and fluctuating water tables (WT), showed significantly lower CH_4 emissions and an increasing trend of CH_4 release associated with a generally increasing WT caused by the progressing rewetting process. Lower N_2O emissions (<24 µg N m^{-2} d^{-1}) were observed at the flooded site. By contrast, the non-flooded peat site with fluctuating WT showed significantly higher N_2O emissions (up to 4178 µg N m^{-2} d^{-1}), in particular at high temperatures during summer time. The present results indicate

Electronic supplementary material The online version of this article (doi:10.1007/s11273-017-9555-x) contains supplementary material, which is available to authorized users.

K. Osterloh · S. Bernsdorf · R. Meissner
Faculty of Natural Sciences III, Institute of Agronomy and Nutritional Sciences, Martin-Luther-University Halle-Wittenberg, Julius-Kühn-Str. 23, 06112 Halle (Saale), Germany

N. Tauchnitz (✉)
Department 25, Agro Ecology and EU Attended Monitoring, State Institute for Agriculture and Horticulture Saxony-Anhalt, Strenzfelder Allee 22, 06406 Bernburg, Germany
e-mail: nadine.tauchnitz@llg.mule.sachsen-anhalt.de

O. Spott
Department of Soil Physics, Helmholtz Centre for Environmental Research-UFZ, Theodor-Lieser-Str. 4, 06120 Halle (Saale), Germany

J. Hepp
Faculty of Natural Sciences III, Institute of Agronomy and Nutritional Sciences, Soil Biogeochemistry, Martin-Luther-University Halle-Wittenberg, Von-Seckendorff-Platz 3, 06120 Halle (Saale), Germany

R. Meissner
Department of Soil Physics, Helmholtz Centre for Environmental Research-UFZ, Lysimeter Station, Falkenberg 55, 39615 Altmärkische Wische, Germany

that permanently flooded conditions during rewetting processes might cause higher CH_4 emissions compared to fluctuating WT which in contrast might enhance N_2O emissions. In total, however, no decreasing trend for CH_4 emissions throughout the five-year renaturation period could be found. At least for N_2O we observed a decreasing trend during rewetting.

Keywords Climate change · Methane · Nitrous oxide · Renaturation · Vegetation shifts · Wetland

Introduction

Natural peatlands represent a tremendous sink of atmospheric carbon dioxide (CO_2) with an accumulation of about 500 ± 100 Gt of the global soil carbon (Yu 2012). Drainage of peatlands for agriculture and forestry use resulted in an accelerated mineralisation of stocked carbon (C) and led to a substantial loss of C and nitrogen in form of CO_2 and nitrous oxide (N_2O), respectively, of globally 2–3 Gt CO_2-eq per year (Joosten and Couwenberg 2009). During the last four decades, rewetting programmes were initialized in Central Europe aiming at reducing greenhouse gas (GHG) emissions and thus, to recover the nutrient sink and ecological functions of pristine peatlands (e.g. Drösler 2005). It has been shown recently that rewetted peatlands may evolve to highly dynamic systems characterised by an increased mobilisation of nutrients, dissolved organic carbon (DOC) and elevated emissions of methane (CH_4) and CO_2 as a result of rapid plant dying and decomposition (Zak et al. 2015). Extremely high CH_4 emissions from rewetted peatlands have been observed and attributed to enhanced C and nutrient availability via the supply of fresh labile organic substrates resulting from the die-off of plants in formerly non-flooded areas (Wilson et al. 2009; Hahn et al. 2015). Afterwards prevailing plant species revealed a major impact on the CH_4 release from rewetted peatlands. First, they are assumed to stimulate CH_4 formation due to root exudation of low-molecular organic compounds from living plants (Hines et al. 2008). Second, they provide freshly reduced C from senescent plant biomass (Lawrence et al. 2013). And third, they permit transport of CH_4 from the anaerobic root zone directly into the atmosphere, by bypassing the aerobic methane-oxidizing peat layers via their well-developed aerenchyma tissue (Whalen 2005). Concomitantly also CH_4 oxidation could occur in the oxygenated zone surrounding the plant roots (Chanton et al. 2002). In addition, the litter quality, in terms of microbial usability of a potential substrate, may differ substantially between plant species (Lai 2009), resulting in differences of CH_4 emissions between various plant species (Zak et al. 2015). It was shown that CH_4 production by decomposition of plant litter with higher nitrogen to C content is supported whereas plant litter composition rich in lignin and polyphenols suppress CH_4 production (Freeman et al. 2001; Thangarajan et al. 2013). Thus, differences in CH_4 emissions between rewetted peat sites can be expected due to differences in plant litter quality of the prevalent plant species.

In contrast to CH_4, rewetted and natural peatlands are generally characterised by low fluxes of N_2O, due to anoxic conditions, which inhibit nitrification processes and cause nitrate limitation at high water saturation (Drösler 2005; Beyer and Höper 2015). Heterotrophic denitrification represents the most important N_2O source in water-logged peat soils (Regina et al. 1996). It was observed previously that N_2O produced from fertilized nitrate via denitrification was stored in the water phase of water-logged peat soils and later on released to the atmosphere only by falling water tables (Aerts and Ludwig 1997; Tauchnitz et al. 2015). As a consequence temporarily strong N_2O bursts can be expected in rewetted peat sites which are characterized by considerably fluctuating water tables. In contrast to CH_4 permanently flooded conditions therefore appear to be at least favourable to decrease N_2O emissions to near zero (Beyer and Höper 2015).

Most rewetting studies in temperate regions were performed in formerly agriculturally used fens with vegetation dominated by hydrophytes. In temperate regions, however, bogs with an oligotrophic vegetation have scarcely been studied so far. Therefore, the present study aimed at quantifying CH_4 and N_2O emissions of a drained transition bog (currently undergoing a rewetting process) over a period of four years by using common closed chamber technique. The main objectives of the study are:

(i) Examining changes in the CH_4 and N_2O release in the course of progressing rewetting,

(ii) Evaluating differences of the CH_4 and N_2O emission patterns between (a) permanently flooded conditions and (b) non-flooded conditions with fluctuating water table (mostly below the peat surface),
(iii) Identifying main drivers of CH_4 and N_2O release at the study site.

Materials and methods

Study site

Field experiments were performed in the German National Park Harz Mountains (central Germany) in a transition bog ecosystem (650–670 m a.s.l) in direct vicinity to the Mt. Brocken (1141 m a.s.l.; 51°48′N, 10°37′E). A detailed description of the study site was previously given by Osterloh et al. (2016). Climate conditions are characterised by a long-term (1961–1990) mean annual temperature of 5.3 °C and a mean annual precipitation of 1278 mm (data from German Weather Service).

Prior to rewetting the study site was drained by several ditches and planted with spruces for forestry use in the 1980s. Rewetting was initiated in 2005 and continued in 2009 and 2010 by closing the ditches and redirecting a small stream into the central peatland area.

For emission analyses two different study sites were chosen within the peatland. One site was characterised by permanently flooded conditions (i.e. water table (WT) above the soil surface since August 2009). Consequently, vegetation changed from grass dominated to sedge-grass-moss dominated structures and plant productivity increased simultaneously.

The second study site was rewetted since June 2010, non-flooded and characterised by a fluctuating WT. The vegetation is characterised by an open young spruce stand rich in *Vaccinium myrtillus* and mosses like *Polytrichum formosum* and *Pleurozium schreberi* but rare in wetland plant species. Rewetting caused a die-off of typical acidic forest plant species followed by a spreading of fen indicator species (namely *Carex canescens, Agrostis canina*, *Molinia caerulea* and *Sphagnum spec*.).

The peat soil layers of both sites are 0.7–1.1 m thick. Dry density is in the range of 0.15–0.22 g cm^3. Both peat sites are characterised by mesotrophic conditions with C:N ratios ranging from 22 to 29. Nitrate contents are low with a maximum of 10 mg g^{-1} dry matter of soil. Detected pH values are in a range of 3.4–3.9 and represent strong acid conditions.

Field sampling and analysis

At each sampling site five polyethylene (PE)-tubes (30 cm diameter, 30 cm height) were pushed 15 cm into the peat soil layer acting as the emission chambers frame. From June 2010 to October 2014 gas release was measured weekly and biweekly (in some cases monthly) by the closed chamber method. Gas measurements were omitted during the winter time (December to February) in case of thick snow covering. For gas sampling, a peat soil column was covered with a dark, opaque PE-chamber (31.5 cm in diameter, 15 cm height) and sealed against the atmosphere by a rubber strip. Chamber then remained closed for two hours while gas samples (aliquot of 100 ml) were transferred by a polypropylene syringe via a three-way valve and a needle from the chamber headspace into a septum-capped vial (20 ml) every 20 min (Tauchnitz et al. 2015). Before gas sampling, all vials were flushed with oxygen to prevent contamination from previous samplings. Gas samples were analysed within a few days for CH_4 and N_2O by a specially configured gas chromatograph (GC 14B, Shimadzu, Japan) equipped with a flame ionization detector (FID for CH_4) and an electron capture detector (ECD for N_2O) (see Segschneider et al. 1996). Emission rates (E) of CH_4 and N_2O (μg m^{-2} h^{-1}) were calculated according to Eq. 1 with ΔC (μg N per norm liter) as the amount of CH_4 and N_2O emitted into a normalized gas volume between $t = 0$ min and $t = 20$, 40, 60, 80, 100 and 120 min, respectively, and h (current champers height) multiplied by 10 and divided by the time in order to correct for the current chamber volume, the covered area and the corresponding closure time. Prior to the calculation of the volume normalized gas concentrations analysed gas concentration was always corrected to standard temperature (i.e. 273 K) and standard pressure (i.e. 1013.25 hPa).

$$E = \Delta C_t \cdot h \cdot 10/t \quad (1)$$

Daily emission rates between the measurement campaigns were obtained by linear interpolation and were used to calculate yearly balances.

Soil temperatures were measured with a puncture thermometer (stainless steel sensor, VWR International GmbH) in 5 and 10 cm depths concomitantly to the gas measurements. In addition to gas sampling WT, redox potential (Eh, Hach Lange GmbH, Germany) as well as oxygen level (O_2, Hach Lange GmbH, Germany) was monitored simultaneously by groundwater piezometers placed 20 cm deep in the peat soil layer adjacent to the chambers. Furthermore, water samples were taken and analysed for ammonium (NH_4^+, photometric, Thermo Electron Corporation, Germany), nitrate, sulfate (NO_3^-, SO_4^{2-}, ion chromatography, Metrohm GmbH & Co. KG Filderstadt, Germany), DOC (Total Carbon Analyser, NPOC method, Shimadzu TOC-V_{CSH}, Japan), iron (Fe, flame atomic absorption spectrometry, Thermo Electron Corporation, Germany) as well as pH (potentiometric, WTW GmbH, Germany) and electric conductivity (EC, conductimetrically, WTW GmbH, Germany). Total N and C contents of plants were determined by high temperature combustion method (Elementar Analysensysteme GmbH, Germany). Total P contents of biomass were analysed by inductively coupled plasma optical emission spectrometry (Ultima 2, Horiba Jobin–Yvon). For analyses of dissolved CH_4 concentration, peat soil water was sampled according to Tauchnitz et al. (2015) and measured by a Headspace GC (HSS 86.50, DANI, Italy) coupled to a GC system (GC 14B, Shimadzu, Japan) equipped with a Porapak Q column (80/100 mesh, 6L, 2 mm ID) and an ECD.

Statistical analyses

Statistical analyses were performed using SPSS (vers. 22). Levene-test was used to examine data sets concerning their equality of variances. Since data were not normally distributed the non-parametric Mann–Whitney-U-test was applied to check for significant differences between peat site locations and years, respectively. Spearman correlation coefficients were calculated to test correlations between gas fluxes and environmental as well as nutrient conditions (e.g. WT, NH_4^+-, NO_3^- -contents, etc.).

Results

Weather and hydrological parameters

Annual precipitation ranged between 974 mm (2011) and 1428 mm (2010) during the study period and was, with the exception of 2010, below the long-term mean (1278 mm) of the study site (Table 1). Annual mean temperature was in a range of 4.6 °C (2010) to 7.3 °C (2014).

Water tables were significantly different between the two sampling sites. At the flooded peat site a permanently high WT with an annual variation of 2.5 cm (2012) to 4.3 cm (2014) above the soil surface was observed. WT at this site showed very low temporal variations (Table 1). In contrast, the recently rewetted non-flooded peat site showed a significantly lower WT (Table 1) with annual means in the range of 11.7 cm (2011) to 1.8 cm (2014) below the soil surface and high temporal variations. WT decreased markedly during summer with lowest value of 33 cm below the soil surface observed at the end of September 2011. In the same year, lowest annual mean WT coincided with the lowest annual precipitation of the whole study period. Nonetheless, in total an increasing trend of the WT was observed for the non-flooded peat site accompanied by decreasing temporal variations, presumably due to progressive rewetting (Table 1).

Characteristics of the peat soil water

Median redox potentials of the flooded and non-flooded peat site were 187 and 259 mV, respectively, with significant differences between both sites (Table 2).

NH_4^+ concentrations and EC of the peat soil water differed significantly between both sites with higher values at the non-flooded site (Table 2). NO_3^- concentrations of the peat soil water were low at both sites with median values of <0.1 mg N L^{-1} and maximal concentrations of 1.5 mg N L^{-1} (observed during strong snow melt coinciding with high precipitation in April 2013; Tables 1, 2). DOC concentrations of the peat soil water (Fig. 1) showed no significant differences between both sites with median values of 21 mg L^{-1} (flooded) and 26 mg L^{-1} (non-flooded). Most notable was a distinct temporal variation in DOC concentrations with highest values during summer.

Table 1 Annual precipitation (P), air temperature (T) and water table (WT) related to soil surface from the flooded and non-flooded peat sites during the study period (June 2010–October 2014); different letters indicate significant differences ($p < 0.05$, $n = 49$, Man-Whitney-U-Test) between years per site (U1) and sites per years (U2)

Year	P (mm)	T (°C)	WT (cm)																		
			Flooded										Non-flooded								
			n	Mean	SD	Min	Q25	Med	Q75	Max	Ul	U2	n	Mean	SD	Min	Q25	Med	Q75	Max	Ul	U2
2010	1428.3	4.6	13	2.9	0.5	1.9	2.7	3.0	3.2	3.5	a	b	12	−3.3	3.2	−8.9	−5.1	−2.9	0.0	0.5	bc	a
2011	973.7	6.8	15	3.0	0.9	1.4	2.4	3.1	3.6	4.2	a	b	16	−11.7	8.4	−32.8	−15.6	−9.4	−6.6	−2.1	a	a
2012	1130.7	5.9	8	2.5	0.8	1.3	1.7	2.7	3.1	3.3	a	b	11	−7.7	7.4	−20.8	−11.1	−5.0	−2.1	−0.3	ac	a
2013	1152.5	5.3	7	3.1	0.5	2.1	3.0	3.3	3.5	3.6	a	b	10	−2.4	3.8	−12.0	−2.4	−1.8	0.2	0.9	b	a
2014	1202.0	7.3	6	4.3	0.5	3.4	4.1	4.3	4.6	4.9	b	b	6	−1.8	1.7	−4.3	−2.6	−1.9	−1.1	0.8	bc	a
	1177.44	6.0	49	3.1								b	56	−5.4								a

SD standard deviation; *Min* Minimum; *Max* Maximum; *Med* median; *Q25* 25%-quantile; *Q75* 75%- quantile

Nutrient contents of biomass

N contents of biomass at the flooded and non-flooded site were in the range of 1.4–3.9% and 1.1–4.1%, respectively (Table S-1). C:N ratios of biomass ranged between 11 and 31 (flooded) and 11 and 43 (non-flooded) and P contents between 0.1 and 0.4% (flooded) and 0.07 and 0.4% (non-flooded). During the study period N and P contents significantly increased, while C:N ratios of biomass decreased at both site.

CH_4 emissions

Highest CH_4 emissions were measured at the permanently flooded peat site ranging from 1.5 to 1195 mg C m^{-2} d^{-1} (Fig. 2; Table S-2).

In addition, a pronounced inter-annual and spatial variation of CH_4 emissions was found. Highest CH_4 release was generally observed during summer associated with high temperatures. Overall, CH_4 emissions at the flooded site showed no significant trend during the whole study period (Fig. 2; Table S-2). Significantly lower CH_4 release was observed at the non-flooded site where emission rates largely tended towards zero. Nonetheless, also higher rates could be detected (up to 571 mg CH_4–C m^{-2} d^{-1}), but mainly at the end of the study period where WT fluctuations diminished and WT remained constantly close the soil surface (Fig. 2).

Mean annual CH_4 emissions of the permanently flooded and non-flooded site accounted for 505 ± 414 kg C ha^{-1} a^{-1} and 68 ± 60 kg C ha^{-1} a^{-1}, respectively.

N_2O emissions

N_2O emissions showed significant differences between both peat sites ($p < 0.05$, $n = 49$, Man-Whitney-U-test). High N_2O release rates of maximal 4178 µg N m^{-2} d^{-1} were found at the non-flooded peat site (Fig. 3; Table S-3) within the first years of the study with highest emissions during summer coinciding with low WT and high temperatures.

Nonetheless, N_2O emissions showed a decreasing trend associated with increasing WT (with exception of the sampling date 25.07.2013). Furthermore, decreasing inter-annual variation of N_2O emission was observed. High N_2O release rates occurred in

Table 2 Peat soil water properties from the flooded and non-flooded peat sites during the study period (June 2010–October 2014), different letters indicate significant differences ($p < 0.05$, Man-Whitney-U-Test) between sites (U)

Parameter	Flooded							Non-flooded						
	n	Mean	SD	Min	Med	Max	U	n	Mean	SD	Min	Med	Max	U
pH (mg L^{-1})	67	4.3	0.4	3.7	4.3	5.3	b	66	4.2	0.4	3.6	4.3	5.2	a
EC (µS cm^{-1})	67	59.1	10.1	40.5	58.0	92.9	a	66	65.9	15.4	47.3	62.1	115.6	b
O_2 (mg L^{-1})	64	3.7	2.7	0.4	2.9	12.1	a	62	3.2	1.5	0.6	3.0	6.4	a
T (°C)	64	8.4	5.0	0.0	9.5	17.6	a	62	8.7	4.7	0.8	9.6	18.6	a
Eh (mV)	61	152	111	−126	187	338	a	60	221	120	−96	259	358	b
SO_4^{2-} (mg L^{-1})	67	7.10	4.48	0.25	6.85	17.62	a	66	6.64	3.14	0.33	6.47	13.21	a
Fe (mg L^{-1})	57	1.93	1.77	0.11	1.15	7.00	a	56	1.61	1.34	0.18	1.10	5.07	a
NO_3–N (mg L^{-1})	67	0.16	0.26	0.00	0.03	1.46	a	66	0.08	0.09	0.00	0.04	0.35	a
NH_4–N (mg L^{-1})	67	0.43	0.33	0.10	0.32	2.05	a	66	0.53	0.35	0.09	0.44	2.20	b
DON (mg L^{-1})	61	0.87	0.64	0.00	0.70	3.03	a	59	0.92	0.67	0.12	0.71	2.89	a
DOC (mg L^{-1})	61	32.18	24.73	4.41	21.07	105.87	a	59	34.75	26.70	4.27	25.91	99.58	a
DOC:DON	60	48	64	9	36	496	a	58	54	82	15	37	622	a

EC electric conductivity; *T* water temperature; *Eh* redox potential; *DON* dissolved organic nitrogen; *DOC* dissolved organic carbon; *SD* standard deviation; *Min* Minimum; *Max* Maximum; *Med* median; *Q*25, 25%-quantile, *Q*75, 75%- quantile

2010, 2011, and 2012 with annual median values of 193, 836 and 415 µg N m^{-2} d^{-1}, respectively. By contrast, markedly lower N_2O loss was observed in the study years 2013 and 2014 which were, however, only significant in 2014 ($p < 0.05$, $n = 49$, Man-Whitney-U-test).

At the permanently flooded peat site N_2O emissions were generally low with a maximal N_2O release of 200 µg N m^{-2} d^{-1}. For 63% of the sampling dates no significant N_2O release (i.e. >24 µg m^{-2} d^{-1}) could be detected, whereas an uptake of N_2O was observed at numerous sampling dates. Contrary to the non-flooded peat site N_2O release of the flooded site showed very low inter-annual and spatial variations. Median annual N_2O values indicate no significant trend throughout the study period (Fig. 3). Annual N_2O emission of the permanently flooded site accounted for -0.03 ± 0.1 kg N ha^{-1} a^{-1}. At the non-flooded site significantly higher emissions of 1.6 ± 1.5 kg N ha^{-1} a^{-1} were found ($p < 0.05$, $n = 4$, Man-Whitney-U-test).

Discussion

CH_4 emission rates and associated drivers

Highest CH_4 emissions of up to 1195 mg C m^{-2} d^{-1} were observed at the permanently flooded peat site (Fig. 2; Table S-2). Similar results for flooded peatlands have been reported elsewhere (Augustin and Joosten 2007; Hahn-Schöfl et al. 2011; Hahn et al. 2015). High emissions are caused by rapid plant die-off and related decomposition processes along with the flooding and thus, providing fresh and easily degradable organic substrates and therefore enhanced C and nutrients availability for methanogenic microorganism (Wilson et al. 2009; Hahn et al. 2015). It was stated elsewhere that the first stage of litter break down during flooding subsequent to die-back of plants resulted in high mass loss of plant nutrient stock (e.g. Aerts and De Caluwe 1997), hydrolysis of polyphenols and other organic substances of mostly low molecular weight, such as carbohydrates and amino acids (Maie et al. 2006). This is supported by the increased mobility of DOC observed during the rewetting process at this site (Osterloh et al. 2016). A positive correlation between CH_4 emissions, which are known as a good proxy for biomass production (Strack et al. 2004), and DOC concentrations in peat soil water were additionally found at the flooded peat site (Table 3).

It was also shown recently that rewetted sites at the studied peatland are characterised by a more unbalanced DOM quality with higher fluorescence and humification indices (HIX), indicating a higher level of aromatic DOM and a stronger humification

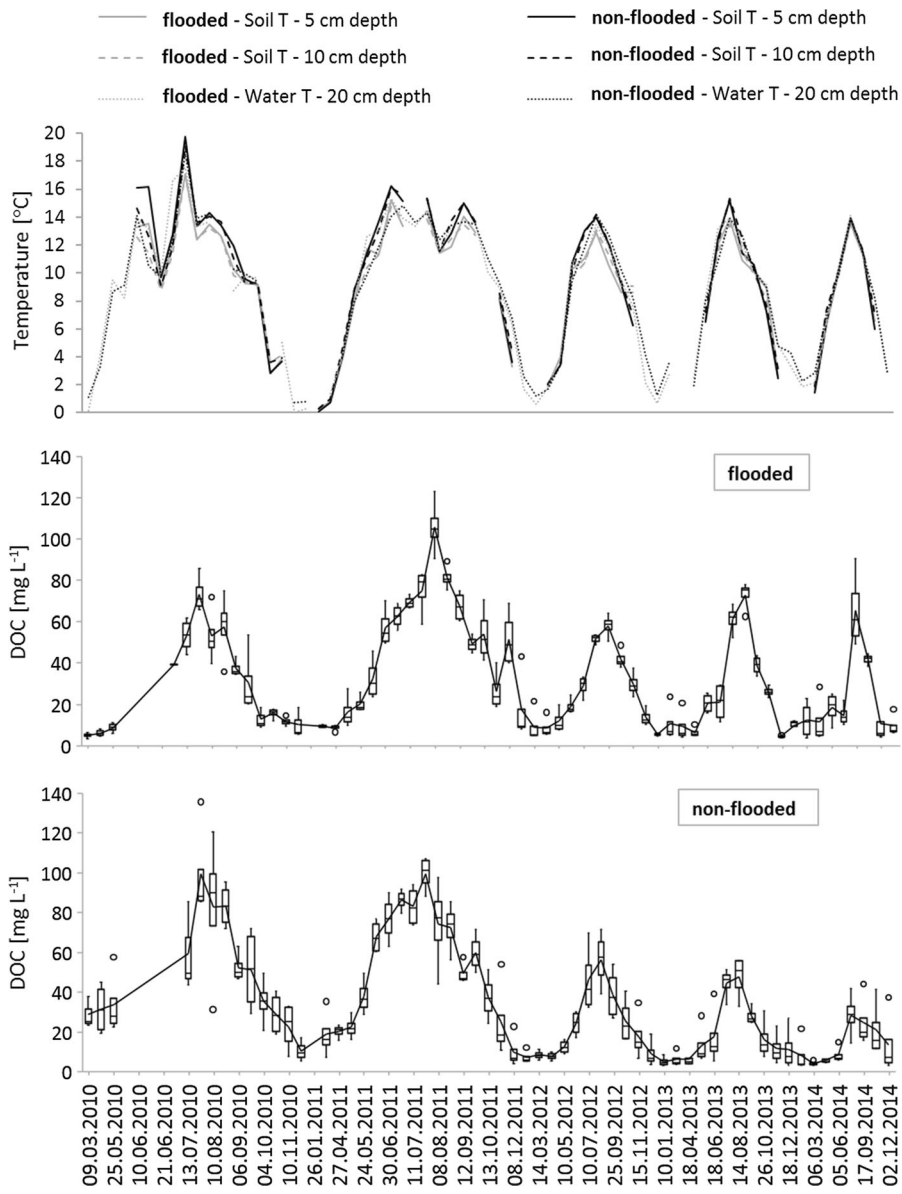

Fig. 1 Temporal course of soil and water temperature (T) and dissolved organic carbon (DOC) from the flooded and non-flooded peat sites (n = 4) during the study period (March 2010–December 2014); within the DOC diagrams *black lines* indicate means, *box* represent 25-, 50 and 75%-quantiles, *dots* are outliers

compared to pristine sites (Herzsprung et al. 2017; Table 4). These findings are well in agreement with Kalbitz et al. (2000), who reported changes of DOM quality during peat degradation with higher amounts of aromatic moieties in degraded peatlands. A high content of aromatic components might contradict with the idea of enhanced C availability via easily degradable organic substrates derived from submerged plants which was previously discussed (e.g. Zak et al. 2015). However rewetted sites are characterised by large seasonal and spatial variations in DOM quality (Herzsprung et al. 2017), indicating a high dynamic of rewetted systems.

A positive correlation was found between CH_4 emissions and EC, as well as DON concentrations of peat soil water at the permanently flooded site, confirming the positive effect of nutrient availability on microbial activity (Table 3). Note that decomposing microorganisms depend on N for their anabolism, thus increasing N concentrations induced by higher litter input suggest enhanced microbial activity (Tremblay and Benner 2006). Therefore, CH_4

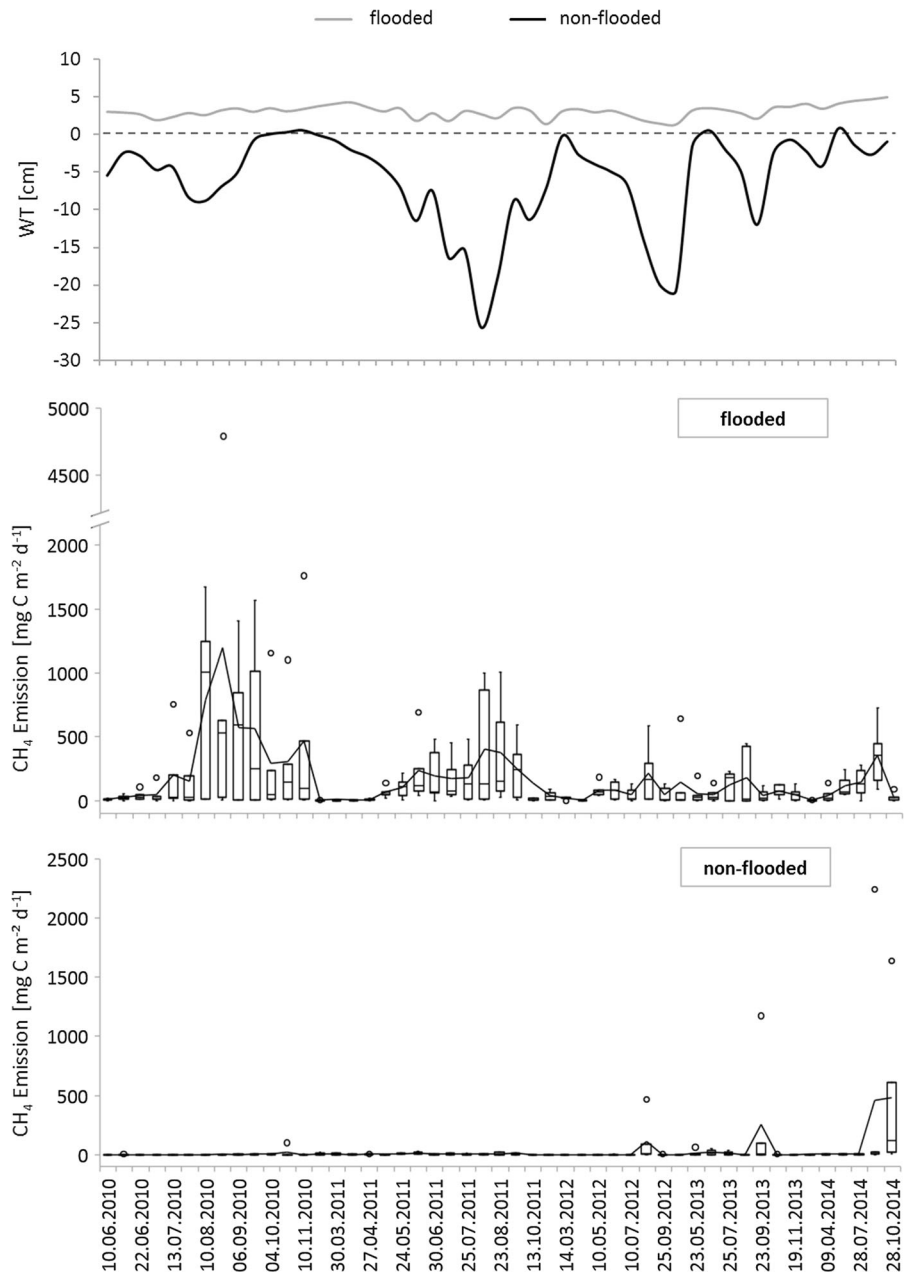

Fig. 2 Temporal course of the CH_4 emissions and water table (WT) from the flooded and non-flooded peat sites (n = 5) during the study period (June 2010–October 2014), *dashed line* in the WT diagram represent soil surface; *black lines* in the emission diagrams indicate means, *box* represent 25-, 50 and 75%-quantiles, *dots* are outliers

production induced by fresh plant litter input during the rewetting processes might differ substantially between various plant species depending on their litter quality (Lai 2009). Zak et al. (2015) obtained a 70-times higher CH_4 production under the hydrophyte *Ceratophyllum demersum* (C:N ratio: 14, N content: 2.8%, P content: 0.68%) compared to *Carex riparia* (C:N ratio: 35, N content: 1.6%; P content: 0.06%). In the present study the C:N ratios, as well as the N and P contents of main plant species at the flooded site (e.g. *Carex canescens*, Table S-1) are in the range of those reported by Zak et al. (2015). Also Bohdálková et al. (2013) presented highest CH_4 emissions from an oligotrophic peat bog in the Czech Republic liked to the occurrence of *Carex*. The high CH_4 production rate under *Carex* species (and other vascular plants) could be attributed to their relatively fast decomposition rates compared to moss species (Lahio 2006). In

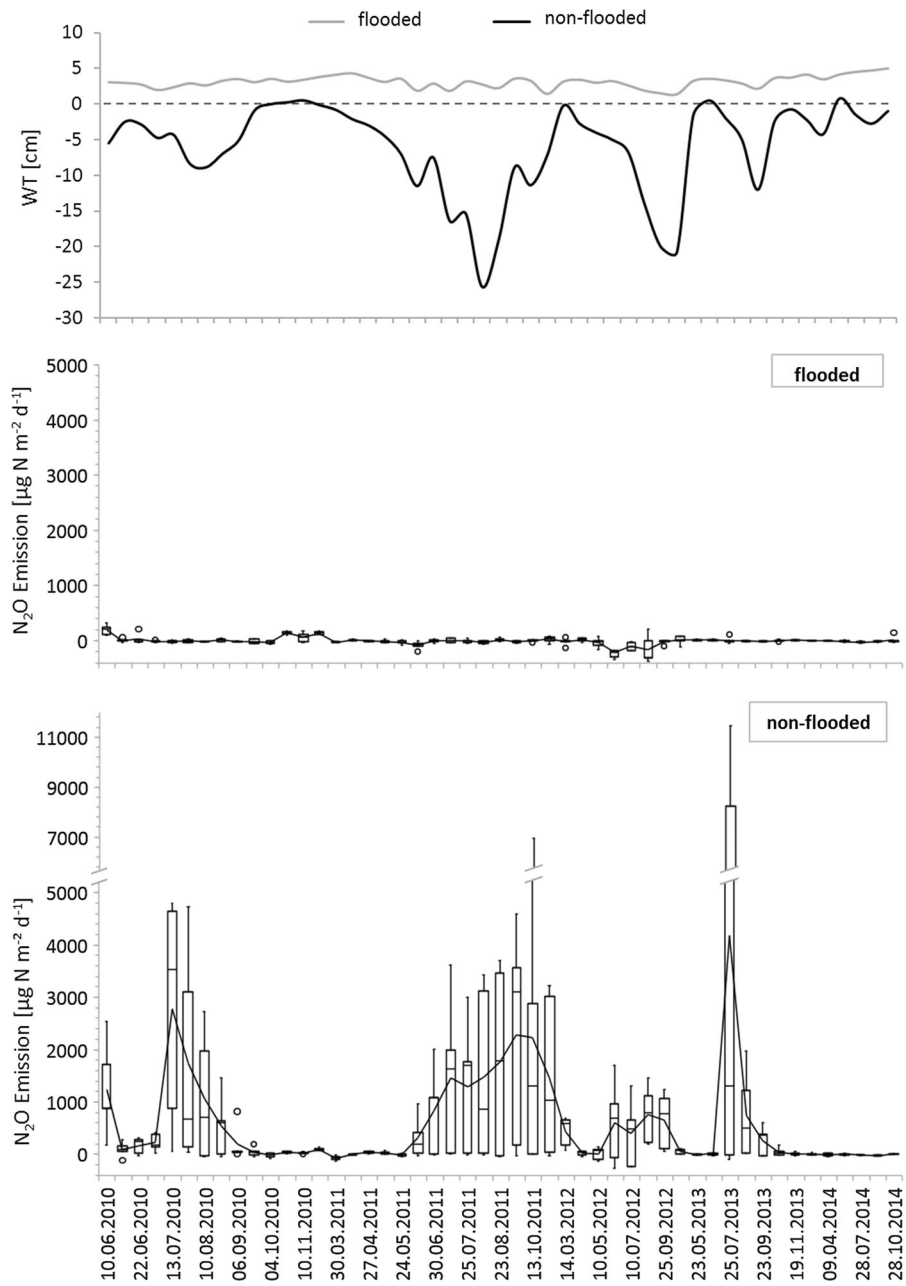

Fig. 3 Temporal course of the N_2O emissions and water table (WT) from the flooded and non-flooded peat sites (n = 5) during the study period (June 2010–October 2014), *dashed line* in the WT diagram represent ground surface; *black lines* in the emission diagrams indicate means, *box* represent 25-, 50 and 75%-quantiles, *dots* are outliers

agreement with other studies (e.g. Aerts and Ludwig 1997) we observed a high spatial variability of CH_4 emissions with coefficients of variation often exceeding 100% within the replicates (Fig. 2; Table S-2). Similar to our results a high inter-annual variation of CH_4 emissions was recognized in recent rewetting studies (Koebsch et al. 2013; Günther et al. 2015). Overall, highest CH_4 release occurred at high temperatures during summer months, as supported by the positive correlation between CH_4 emissions and soil temperature at the permanently flooded site. Also Audet et al. (2013) and Beyer and Höper (2015) found increased CH_4 production rates under warmer conditions. Additionally, a significant negative correlation between CH_4 emissions and SO_4^{2-} concentrations of peat soil water was observed at the flooded peat site (Table 3). This finding confirms previously published results by Audet et al. (2013). CH_4 is produced under

Table 3 Spearman correlation coefficients between methane (CH_4) and nitrous oxide (N_2O) emissions from the flooded and non-flooded peat sites and relevant parameters

Parameter	Spearman correlation coefficients			
	N_2O (μg m^{-2} d^{-1})		CH_4 (μg m^{-2} d^{-1})	
	Flooded	Non-flooded	Flooded	Non-flooded
WT (cm)	0.25	−0.63**	−0.26	0.11
Soil-$T_{5\ cm}$ (°C)	−0.27	0.49**	0.51**	−0.01
Soil-$T_{10\ cm}$ (°C)	−0.29*	0.54**	0.56**	0.04
Water-$T_{20\ cm}$ (°C)	−0.27	0.54**	0.53**	0.08
O_2 (mg L^{-1})	0.34*	−0.35*	−0.60**	−0.02
Eh (mV)	0.24	0.07	−0.37**	−0.26
pH	0.08	−0.45**	−0.47**	0.26
EC (μS cm^{-1})	0.00	0.27	0.37**	−0.40**
SO_4^{2-} (mg L^{-1})	0.34*	−0.39**	−0.74**	−0.10
Fe (mg L^{-1})	−0.17	0.63**	0.51**	0.05
NO_3–N (mg L^{-1})	0.37**	−0.09	−0.26	−0.14
NH_4–N (mg L^{-1})	0.23	0.26	−0.22	−0.09
DON (mg L^{-1})	−0.15	0.66**	0.45**	0.09
DOC (mg L^{-1})	−0.19	0.67**	0.56**	0.04
Date	−0.15	−0.27	−0.14	0.45**

WT water table related to soil surface; *Soil-/Water-T…$_{cm}$* soil/water temperature in… cm depth; *Eh* redox potential; *EC* electric conductivity; *DON* dissolved organic nitrogen; *DOC* dissolved organic carbon

* $p < 0.05$; ** $p < 0.01$

Table 4 Humification index (HIX) and humic-like fluorescence of peat soil water in 20 cm depth of pristine and rewetted peat sites during sampling time of May 2013–April 2014; different letters indicate significant differences ($p < 0.05$, Man-Whitney-U-Test) between sites (U)

Study sites	n	HIX				Humic-like fluorescence			
		Med	Min	Max	U	Med	Min	Max	U
All values									
Pristine site	10	6.1	4.7	6.9	a	9797	8978	12031	a
Rewetted site	40	11.6	4.0	23.9	b	8081	2295	30544	a
Summer values (May–October)									
Pristine site	5	5.9	4.7	6.8	a	9981	9185	12031	a
Rewetted site	20	13.2	9.8	23.9	b	15873	7406	30544	a
Winter values (November–April)									
Pristine site	5	6.2	5.7	6.5	a	9613	8978	10527	a
Rewetted site	20	9.0	4.0	13.6	a	6111	2295	10190	a

Med median; *Min* minimum; *Max* maximum

strictly anaerobic conditions by methanogenic *archaea* species as a result of fermentation of organic compounds (acetate) or reduction of CO_2 by H_2 (Le Mer and Roger 2001). The negative effect of SO_4^{2-} can be explained by the competition between SO_4^{2-} reducing bacteria, using SO_4^{2-} as an electron acceptor, and methanogens regarding the utilization of H_2 and acetate (Conrad 2007). Note that methanogenesis might be also inhibited in the presence of NO_3^- and iron (Fe(III)) as electron acceptors (Le Mer and Roger 2001). However, we could not confirm this latter suggestion. Contradicting, a positive correlation between CH_4 emissions and Fe concentrations of peat soil water was found, which was assumed rather a result of the interaction between DOC and Fe due to the formation of chelate complexes (Osterloh et al. 2016).

Significantly lower CH_4 emissions were detected at the non-flooded peat site with fluctuating WT compared to the permanently flooded site (Fig. 2). Note that the WT level is one of the key factors of CH_4 emissions in peatlands determining the thickness of the aerobic soil layer where CH_4 oxidation takes place (Lai 2009; Wilson et al. 2009). In contrast to Jungkunst et al. (2008) who found a non-linear correlation between WT position and CH_4 emissions, no significant correlation was detectable in the present

study at the non-flooded peat site (Table 3). Nevertheless, higher CH_4 emissions coincide with higher WT throughout the whole study period (Fig. 2). Differences between the flooded and non-flooded peat site were found to be concerning CH_4 concentrations within the peat soil water (Table 5). In addition, no significant correlation between temperature and relevant hydro-chemical parameters to CH_4 emission could be seen at the non-flooded site with fluctuating WT, in contrast to the permanently flooded site (Table 3), supporting the assumption of differences in CH_4 production conditions between the both sites. A very effective way to channel large amounts of CH_4 from the anaerobic zone directly into the atmosphere, i.e. bypassing the aerobic methane-oxidizing peat layers, is a plant mediated transport through the aerenchyma of aquatic vascular plants (Whalen 2005). However, such an effect, mainly induced by wetland plants (reed plant species), are not relevant for the here presented data due to their absence on both study sites.

In 2010 (beginning of the study period), the non-flooded peat site showed significantly lower CH_4 concentrations in depths of 30 and 60 cm compared to the permanently flooded peat site. Already Chanton and Whiting (1995) described that at high rates of methanogenesis supersaturation of CH_4 might occur in the porewater of deeper anaerobic peat layers and hence, gas bubbles are formed when the partial pressure of all dissolved gases in solution exceeds the hydrostatic pressure in soil water. These bubbles accumulate within the peat profile and can be released suddenly in case of a rapid drop of atmospheric pressure, a lowering of hydrostatic pressure (e.g. via decreasing WT), a rise in temperature or mechanical stress by e.g. walking on the peat surface (Tokida et al. 2007). It is well known that the process of ebullition may contribute significantly to the total CH_4 fluxes in peatlands (Tokida et al. 2007). It can be assumed that the high inter-annual and spatial variation of CH_4 emissions observed in our study largely resulted from ebullition events.

N_2O emissions rates and associated drivers

Generally, low N_2O emissions were observed at the permanently flooded peat site, which is in line with results from previous studies reporting negligible N_2O release from rewetted peatlands (Drösler 2005; Beyer and Höper 2015; Günther et al. 2015). Note that heterotrophic denitrification should be the major process of N_2O formation in peatlands (Regina et al. 1996). In fact, a weak positive correlation between N_2O release at the flooded peat site and NO_3^- concentration in the peat soil water was observed indicating that N_2O formation mainly depended on the availability of NO_3^- as a source for heterotrophic denitrification (Table 3). It could therefore be assumed that the process of denitrification is still active at this peat site, even at very low soil pH, as long as NO_3^- is available (Tauchnitz et al. 2015). Actually Rückauf et al. (2004) stated that N_2 should be the predominant product of denitrification in water-saturated peatlands. This could explain the extremely low N_2O release rates observed at the permanently flooded peat site. The permanent flooding could limit the mobility of N_2O in the sense of moving towards the atmosphere and hence N_2 production is favoured (i.e. accumulation and consumption of N_2O) over N_2O release (Jørgensen et al. 2012). Previous studies already revealed an accumulation of N_2O in peat soil

Table 5 Dissolved methane (CH_4) concentrations in peat soil water in 30 and 60 cm depth from the flooded and non-flooded peat sites during sampling time of September 2010; different letters indicate significant differences ($p < 0.05$, Man-Whitney-U-Test) between sites per depth (U)

Sampling depth (cm)	CH_4–C µg L^{-1}															
	Flooded								Non-flooded							
	n	Mean	SD	Min	Q25	Med	Q75	Max	n	Mean	SD	Min	Q25	Med	Q75	Max
30 U	27	4730 b	1563	1845	3322	4828	5896	7422	19	204 a	74	61	149	205	255	337
60 U	27	4374 b	1486	1942	3227	4338	5248	7617	19	232 a	135	21	126	194	300	516

SD standard deviation; *Min* minimum; *Max* maximum; *Med* median; *Q*25, 25%-quantile; *Q*75, 75%- quantile

water after NO_3^- addition, which was then released to the atmosphere only at falling WT (e.g. Tauchnitz et al. 2015). Conversely, Krüger et al. (2014) found a positive response of rising WT on N_2O emission during rewetting of a temporally dry wetland in Southern Africa. These high N_2O emissions after short-term changes in WT were explained by a flushing out of N_2O which had accumulated in deeper zones. In contrast to the flooded site significantly higher N_2O emission rates were observed at the non-flooded site associated with lower WT and considerable WT fluctuations (Table 1). Similar results concerning a significant response of N_2O release on WT fluctuations were reported previously (Aerts and Ludwig 1997; Tauchnitz et al. 2015). It was pointed out that N_2O fluxes of water-logged soils are (besides the availability of NO_3^-) mainly controlled by the balance of production and consumption of N_2O and by convective or diffusive transport (e.g. Well et al. 2003). As mentioned above N_2O accumulates in the saturated zone when NO_3^- was added to a rewetted peat soil. This accumulation then occurs under constant high WT conditions due to a transport limitation towards the atmosphere. A falling WT leads then to a release of accumulated N_2O based on a significant decrease of diffusional transport limitations, i.e. a direct connection of the soil gas phase to the atmosphere (e.g. Tauchnitz et al. 2015). A significant negative correlation between WT and N_2O emissions was detected at the non-flooded peat site (Table 3), which proves the impact of falling WT on N_2O release. However it is also known that N_2O emission of water-logged soils can be promoted by plants through rapid gas phase diffusion in the aerenchyma of wetland plants (reed plant species) (Jørgensen et al. 2012). But such species were not present at our study site.

In addition to the above mentioned negative correlation between WT and N_2O emissions, a positive relationship between soil (and water) temperature and N_2O emissions was found for the non-flooded peat site (Table 3). It is known from other studies (e.g. Silvan et al. 2002) that microbial processes (such as denitrification) are favoured by warm conditions and falling WT. This framework may support NO_3^- formation via nitrification processes, causing high N_2O release during summer at simultaneously wet conditions. However, increased N_2O emissions during freeze–thaw-cycles as shown e.g. by Jungkunst et al. (2006), could not be confirmed here. Moreover, a significant positive correlation between N_2O release and DOC concentrations of the peat soil water was observed at the non-flooded peat site (Table 3). Since most denitrifying bacteria couple organic C oxidation and NO_3^- oxidation to gain energy (Knowles 1982), it is obvious that a high supply of readily biodegradable organic C would favour denitrification. The impact of a high availability of organic C on N_2O release was already shown in other studies (e.g. Eickenscheidt et al. 2014). The positive relationship between N_2O emission and dissolved organic N (DON) concentrations of peat soil water at the non-flooded peat site furthermore indicates a stimulation of microbial activity due to increasing amounts bioavailable N (Glatzel et al. 2006).

Nitrate contents of the peat soils at both peat sites were low and showed no significant differences among each other. Furthermore no differences in NO_3^- contents of peat soil and peat soil water during the study period could be identified (unpublished data), which may indicate an enhanced NO_3^- availability via nitrification. We therefore suggest that NO_3^- mainly originates from atmospheric N deposition. This assumption is supported by the atmospheric NO_3^- deposition of 18 kg N ha^{-1} a^{-1} observed at the study site (Tauchnitz et al. 2010). Hill et al. (2016) also found a positive correlation between potential denitrification and atmospheric NO_3^- input for a bog watershed. Another pathway might be the NO_3^- transport via slope water into the bog (Tauchnitz et al. 2010).

Evolution of CH_4 and N_2O emission in the course of rewetting

The annual CH_4 emission accounted for 68 ± 60 and 505 ± 414 kg C ha^{-1} a^{-1} at the non-flooded and flooded peat site, respectively. Thus, a global warming potential over 100 years (GWP100) of 1428 (non-flooded site) and 10605 kg CO_2-eq. ha^{-1} a^{-1} (flooded site) can be derived considering a 21-fold higher global warming potential of CH_4 compared to CO_2 (IPCC 1996). The results are in line with previous studies of rewetted temperate peatlands showing a wide range of −0.75 to 3525 kg C ha^{-1} a^{-1} (Meyer et al. 2001; Hahn et al. 2015) with the highest CH_4 release observed in a flooded coastal fen in Northeast Germany (Hahn et al. 2015).

At the permanently flooded peat site CH_4 release remained high during whole study period but revealed

a small decreasing trend within five years after flooding. Similar high CH_4 emissions, recently after flooding were found in fens in Northeast and Northwest Germany where CH_4 was presumably emitted from a newly formed litter layer derived by dead plant residues (Meyer et al. 2001; Günther et al. 2015; Hahn et al. 2015). In contrast, Drösler (2005) showed markedly lower CH_4 fluxes of a recently flooded bog heathland in South Germany which in part remained below common CH_4 emissions of natural bog sites. After Höper et al. (2008) degradation of fen mires after flooding occurs in three different phases with different characteristics. The first initial phase is characterised by extremely high CH_4 emissions associated with a low net CO_2 uptake, resulting in a negative climate effect. In the second phase a strong reduction of CH_4 emissions occurs accompanied by an increase of CO_2 uptake and thus, a slightly positive climate effect. Whereas in the third phase low CH_4 release and low net CO_2 uptakes can be expected (Asaeda et al. 2002). Until yet little is known about the duration of those phases because most rewetting studies are based on data gathering shortly after rewetting. Beyer and Höper (2015) observed high CH_4 emissions with interannual variations at two flooded sites of a bog in Northwest Germany even 30 years after rewetting. Drösler (2005) reported that after 10 years of renaturation the optimal conditions for low greenhouse gas emissions had not been achieved. Besides that, Günther et al. (2014) found that after 15 years of rewetting the net greenhouse gas balance of a fen in Northeast Germany was similar to those of pristine fens. Note that the present study represents the initial phase of rewetting.

In contrast to the permanently flooded peat site, a small increasing trend of CH_4 emissions was observed at the non-flooded site with fluctuating WT, associated with an increasing WT and observed dieback of plants, which was also reported by Hahn et al. (2015).

Different approaches were discussed to mitigate CH_4 emissions of flooded fen peatlands. When keeping the WT on a constant level, with a threshold value for the mean annual water level of about 10 cm below the soil surface, high CH_4 emissions during rewetting could be prevented (Höper et al. 2008). Other wetland studies have demonstrated that an aerobic layer of 20–30 cm is sufficient to ensure low CH_4 emissions (Koebsch et al. 2013) which are, however, unsuitable for establishing *Sphagnum* species. Zak et al. (2015) suggested the removal of upper degraded peat layers in order to minimize nutrient availability. As another possibility with the same goal biomass harvesting, which could be used then for energetic purposes, was proposed (Günther et al. 2015).

The annual N_2O emissions of the permanently flooded and non-flooded peat site accounted for -0.03 ± 0.1 and 1.6 ± 1.5 kg N ha^{-1} a^{-1}, respectively. Thus, a GWP100 of -9.3 and 496 kg CO_2-eq. ha^{-1} a^{-1} can be expected when considering a 310-fold higher GWP of N_2O compared to CO_2 (IPCC 1996). The results are in line with previous studies of restored temperate peatlands showing a range of -0.42 to 3.2 kg N ha^{-1} a^{-1} (e.g. Meyer et al. 2001; Beyer and Höper 2015) with highest N_2O release reported for rewetted fens.

At the permanently flooded peat site no significant trend in N_2O emission was observed during the whole study period (five years after rewetting started). Gas measurements started one year after rewetting already at permanent water saturation, which resulted in negligible N_2O emissions. These results were confirmed by other rewetting studies who reported very low N_2O fluxes and even a net sink function of flooded peatlands (Meyer et al. 2001; Drösler 2005; Günther et al. 2015). The non-flooded peat site with fluctuating WT also showed no significant trend in N_2O emission throughout the study period. Nevertheless, annual differences with significantly lower N_2O release rates towards the end of study could be observed, most likely due to less intensive WT fluctuations and generally higher WT levels. Particularly in the first two years, parallel to the progressing rewetting, N_2O emissions reached their maximum (up to 4178 µg N m^{-2} d^{-1}), associated with a significantly decreasing WT during summer. These high emissions did not occur as strong three years after rewetting. However, one significant N_2O outlier with a release rate of 11454 µg N m^{-2} d^{-1} could be observed on 25th of July 2013 (Fig. 3), which was presumably caused by optimal conditions for N_2O formation, i.e. high soil temperatures (15.2 °C in 5 cm depth) combined with a WT just slightly below the peat soil surface (−2 cm) and a weak precipitation event (4 mm) shortly before gas measurements.

A weak negative trend of N_2O emissions at the non-flooded peat site with lower and fluctuating WT was observed since 2012 (Fig. 3). Drösler (2005) showed

lower N_2O release rates of maximal 672 µg N m^{-2} d^{-1} at a bog in South Germany flooded parallel to the gas measurements. However, the mean water table was thereby 44.5 cm above the peat soil surface with low WT fluctuations. Meyer et al. (2001) reported N_2O emissions near zero and even an uptake of N_2O at a recently flooded (parallel to gas measurements) shallow grassland fen in Northwest Germany with a maintained WT of −10 cm. At an also lately, but less rewetted site (WT −30 cm below soil surface) they observed slightly higher N_2O emissions. Also Augustin and Joosten (2007) described a rewetted grassland fen in Northeast Germany as a weak source of N_2O (0.7 kg N ha^{-1} a^{-1}). In general, it is well known that frequently alternating wet-dry-cycles may enhance N_2O emissions (e.g. Harrison-Kirk et al. 2013). Concerning the impact of N_2O emissions during rewetting a fast adjustment of permanent high WT near the peat soil surface (i.e. a permanent flooding) should be preferred. This will have, however, negative consequences with respect to CH_4 release which will then significantly increase.

Conclusions

The present study showed an increasing trend of CH_4 release and decreasing N_2O emissions during rewetting. Permanent flooded conditions caused the highest CH_4 release. Thus, the present results reveal that not only flooded fens with high nutrient availability of decomposed plant litter but also bog can contribute to very high CH_4 emissions when kept under permanently flooded conditions. Approaches to mitigate CH_4 emissions from rewetted peatlands as discussed in literature (e.g. removal of upper degraded peat soil layer to minimize nutrient availability or biomass harvesting) are not realizable in nature reserves as studied here. Also the maintenance of a constant WT near 10 cm below the peat soil surface appears to be practically unfeasible due to spatial and seasonal variations at the large scale.

The main objective in the presented rewetting program was the recovery of ecological functions by establishing peat forming plants (*Sphagnum*) and hence, the long-term C sink function of the peatland. Present results have revealed that five years after rewetting no decreasing trend of CH_4 emissions were detected. It could be expected that GHG emissions will decrease in the long term after rewetting. However, in order to generate a better understanding of the functionality of restored peatlands and to improve the planning process of rewetting approaches long-term GHG emission monitoring and the development of vegetation patterns after rewetting are required. Furthermore, the impact of vegetation structure and quality of dissolved organic carbon on CH_4 release should be addressed more in detail in further research.

Acknowledgements The authors gratefully acknowledge the Federal State of Saxony-Anhalt under the European Agricultural Fund for the Rural Development (ELER) and the German Research Foundation (DFG) for funding the research. A special thanks goes to the German National Park Harz Mountains as well as Mrs. R. Nauendorf and Mr. M. Engel† (Martin-Luther-University Halle-Wittenberg) for supporting the fieldwork and to Mr. B. Apelt (Helmholtz Centre for Environmental Research – UFZ) for conducting the gas analysis.

Funding The research was funded by the Federal State of Saxony-Anhalt under the European Agricultural Fund for the Rural Development (ELER) and by the German Research Foundation (DFG), Deutsche Forschungsgemeinschaft.

References

Aerts R, de Caluwe H (1997) Initial litter respiration as indicator for long-term leaf litter decomposition of Carex species. Oikos 80(2):353–361. doi:10.2307/3546603

Aerts R, Ludwig F (1997) Water-table changes and nutritional status affect trace gas emissions from laboratory columns of peatland soils. Soil Biol Biochem 29(11/12):1691–1698

Asaeda T, Nam LH, Hietz P, Tanaka N, Karunaratne S (2002) Seasonal fluctuations in live and dead biomass of *Phragmites australis* as described by a growth and decomposition model: implications of duration of aerobic conditions for litter mineralization and sedimentation. Aquat Bot 73(3):223–239

Audet J, Elsgaard L, Kjaergaard C, Larsen SE, Hoffmann CC (2013) Greenhouse gas emissions from a Danish riparian wetland before and after restoration. Ecol Eng 57:170–182

Augustin J, Joosten H (2007) Peatland rewetting and the greenhouse effect. International Mire Conservation Group Newsletter 2007/3:29–30

Beyer C, Höper H (2015) Greenhouse gas exchange of rewetted bog peat extraction sites and a *Sphagnum* cultivation site in northwest Germany. Biogeosciences 12:2101–2117. doi:10.5194/bg-12-2101-2015

Bohdálková L, Čuřík J, Kuběna AA, Bůzek F (2013) Dynamics of methane fluxes from two peat bogs in the Ore Mountains, Czech Republic. Plant Soil Environ 1:14–21

Chanton JP, Whiting GJ (1995) Trace gas exchange in freshwater and coastal marine environments: ebullition and transport by plants. In: Matson PA, Harriss RC (eds) Biogenic trace gases: measuring emissions from soil and water. Blackwell, Oxford, pp 98–125

Chanton JP, Arkebauer TJ, Harden HS, Verma SB (2002) Diel variation in lacunal CH_4 and CO_2 concentration and δ13C in Phragmites australis. Biogeochemistry 59(3):287–301. doi:10.1023/A:1016067610783

Conrad R (2007) Microbial ecology of methanogens and methanotrophs. Adv Agron 96:1–63

Drösler M (2005) Trace gas exchange and climatic relevance of bog ecosystems, southern Germany. Dissertation, Technische Universität München

Eickenscheidt T, Heinichen J, Augustin J, Freibauer A, Drösler M (2014) Nitrogen mineralization and gaseous nitrogen losses from waterlogged and drained organic soils in a black alder (*Alnus glutinosa* (L.) Gaertn.) forest. Biogeosciences 11:2961–2976. doi:10.5194/bg-11-2961-2014

Freeman C, Ostle N, Kan H (2001) An enzymic latch on a global carbon store. Nature 409:149

Glatzel S, Lemke S, Gerold G (2006) Short-term effects of an exceptionally hot and dry summer on decomposition of surface peat in a restored temperate bog. Eur J Soil Biol 42:219–229

Günther AB, Huth V, Jurasinski G, Glatzel S (2014) Scale-dependent temporal variation in determining the methane balance of a temperate fen. Greenh Gas Meas Manag 4:41–48. doi:10.1080/20430779.2013.850395

Günther AB, Huth V, Jurasinski G, Glatzel S (2015) The effect of biomass harvesting on greenhouse gas emissions from a rewetted temperate fen. GCB Bioenerg 7:1092–1106. doi:10.1111/gcbb.12214

Hahn J, Köhler S, Glatzel S, Jurasinski G (2015) Methane exchange in a coastal fen in the first year after flooding—a systems shift. PLoS ONE 10(10):e0140657. doi:10.1371/journal.pone.0140657

Hahn-Schöfl M, Zak D, Minke M, Gelbrecht J, Augustin J, Freibauer A (2011) Organic sediment formed during inundation of a degraded fen grassland emits large fluxes of CH_4 and CO_2. Biogeosciences 8:1539–1550. doi:10.5194/bg-8-1539-2011

Harrison-Kirk T, Beare MH, Meenken ED, Condron LM (2013) Soil organic matter and texture affect responses to dry/wet cycles: effects on carbon dioxide and nitrous oxide emissions. Soil Biol Biochem 57:43–55. doi:10.1016/j.soilbio.2012.10.008

Herzsprung P, Osterloh K, von Tümpling W, Harir M, Hertkorn N, Schmitt-Kopplin P, Meissner R, Bernsdorf S, Friese K (2017) Differences in DOM of rewetted and natural peatlands—results from high-field FT-ICR-MS and bulk optical parameters. Sci Total Environ. doi:10.1016/j.scitotenv.2017.02.054

Hill BH, Jicha TM, Lehto LLP, Elonen CM, Sebestyen SD, Kolka RK (2016) Comparisons of soil nitrogen mass balances for an ombrotrophic bog and a minerotrophic fen in northern Minnesota. Sci Total Environ 550:880–892

Hines ME, Duddleston KN, Rooney-Varga JN, Fields D, Chanton JP (2008) Uncoupling of acetate degradation from methane formation in Alaskan wetlands: connections to vegetation distribution. Glob Biogeochem Cycle 22:GB2017

Höper H, Augustin J, Cagampan J, Drösler M, Lundin L et al (2008) Restoration of peatlands and greenhouse gas balances. In: International Peat Society (ed) Peatlands and climate change. International Peat Society, Jyväskylä, pp 182–210

IPCC (1996) Climate change, the science of climate change, contribution of working Group I to the second assessment report of the intergovernmental panel on climate change. Cambridge University Press, Cambridge

Joosten H, Couwenberg J (2009) Are emission reductions from peatlands MRV-able? Report. Wetlands International, Wageningen, p 14

Jørgensen CJ, Struwe S, Elberling B (2012) Temporal trends in N_2O flux dynamics in a Danish wetland—effects of plant-mediated gas transport of N_2O and O_2 following changes in water level and soil mineral-N availability. Glob Chang Biol 18:210–222

Jungkunst HF, Freibauer A, Neufeldt H, Bareth G (2006) Nitrous oxide emissions from agricultural land use in Germany—a synthesis of available annual field data. J Plant Nutr Soil Sci 169:341–351. doi:10.1002/jpln.2005219542006

Jungkunst HF, Flessa H, Scherber C, Fiedler S (2008) Groundwater level controls CO_2, N_2O and CH_4 fluxes of three different hydromorphic soil types of a temperate forest ecosystem. Soil Biol Biochem 40:2047–2054

Kalbitz K, Solinger S, Park JH, Michalzik B, Matzner E (2000) Controls on the dynamics of dissolved organic matter in soils: a review. Soil Sci 165:277–304

Knowles R (1982) Denitrification. Microbiol Rev 46(1):43–70

Koebsch F, Glatzel S, Jurasinski G (2013) Vegetation controls methane emissions in a coastal brackish fen. Wetl Ecol Manag 21(5):323–337

Krüger JP, Beckedahl H, Gerold G, Jungkunst HF (2014) Greenhouse gas emission peaks following natural rewetting of two wetlands in the southern Ukhahlamba-Drakensberg Park, South Africa. S Afr Geogr J 96(2):113–118. doi:10.1080/03736245.2013.847798

Lahio R (2006) Decomposition in peatlands: reconciling seemingly contrasting results on the impacts of lowered water levels. Soil Biol Biochem 38:2011–2024

Lai DYF (2009) Methane dynamics in northern peatlands: a review. Pedosphere 19:409–421

Lawrence BA, Jackson RD, Kucharik CJ (2013) Testing the stability of carbon pools stored in tussock sedge meadows. Appl Soil Ecol 71:48–57

Le Mer J, Roger P (2001) Production, oxidation, emission and consumption of methane by soils: a review. Eur J Soil Biol 37:25–50

Maie N, Jaffé R, Miyoshi T, Childers D (2006) Quantitative and qualitative aspects of dissolved organic carbon leached from senescent plants in an oligotrophic wetland. Biogeochemistry 78:285–314

Meyer K, Höper H, Blankenburg J (2001) Spurengashaushalt und Klimabilanz von Niedermooren unter dem Einfluß des Vernässungsmanagements. In: Kratz R, Pfadenhauer J (eds) Ökosystemmanagement für Niedermoore, Strategien und Verfahren zur Renaturierung. Ulmer, Stuttgart, pp 104–111

Osterloh K, Tauchnitz N, Wehner I, Bernsdorf S, Kison H-U, Meissner R (2016) Impact of peatland rewetting in the German National Park Harz Mountains on the mobility of dissolved organic carbon (DOC). WasserWirtschaft 106(2/3):20–27

Regina K, Nykanen H, Silvola J, Martikainen PJ (1996) Fluxes of nitrous oxide from boreal peatlands as affected by peatland type, water table level and nitrification capacity. Biogeochemistry 35:401–418

Rückauf U, Augustin J, Russow R, Merbach W (2004) Nitrate removal from drained and reflooded fen soils affected by soil N transformation processes and plant uptake. Soil Biol Biochem 36:77–90

Segschneider H-J, Sich I, Russow R (1996) Use of special configured gas chromatographic system for the simultaneous determination of methane, nitrous oxide and carbon dioxide in ambient air and soil atmosphere. Transactions of the 9th nitrogen workshop, Braunschweig, September 1996, pp. 547–550

Silvan N, Regina K, Kitunen V, Vasander H, Laine J (2002) Gaseous nitrogen loss from a restored peatland buffer zone. Soil Biol Biochem 34(5):721–728

Strack M, Waddington JM, Tuittila E-S (2004) Effect of water table drawdown on northern peatland methane dynamics: implications for climate change. Glob Biogeochem Cycles. doi:10.1029/2003GB002209

Tauchnitz N, Meissner R, Bernsdorf S, Wegener U (2010) Nitrogen fluxes of a slope mire in the German Harz Mountains. Water Air Soil Pollut 205:107–112. doi:10.1007/s11270-009-0059-z

Tauchnitz N, Spott O, Russow R, Bernsdorf S, Glaser B, Meissner R (2015) Release of nitrous oxide and dinitrogen from a transition bog under drained and rewetted conditions due to denitrification: results from a 15 N nitrate-bromide double-tracer study. Isot Environ Health Stud 51:300–321

Thangarajan R, Bolan NS, Tian G, Naidu R, Kunhikrishnan A (2013) Role of organic amendment application on greenhouse gas emission from soil. Sci Total Environ 465:72–96

Tokida T, Mizoguchi M, Miyazaki T, Kagemoto A, Nagata O, Hatano R (2007) Episodic release of methane bubbles from peatland during spring thaw. Chemosphere 70:165–171

Tremblay L, Benner R (2006) Microbial contributions to N-immobilization and organic matter preservation in decaying plant detritus. Geochim Cosmochim Acta 70:133–146

Well R, Augustin J, Meyer K, Myrold DD (2003) Comparison of field and laboratory measurement of denitrification and N_2O production in the saturated zone of hydromorphic soils. Soil Biol Biochem 35:783–799

Whalen SC (2005) Biogeochemistry of methane exchange between natural wetlands and the atmosphere. Environ Eng Sci 22:73–94

Wilson D, Alm J, Laine J, Byrne KA, Farrel EP, Tuittila E-S (2009) Rewetting of cutaway peatlands: are we re-creating hot spots of methane emissions? Restor Ecol 17:796–806

Yu ZC (2012) Northern peatland carbon stocks and dynamics: a review. Biogeosciences 9:4071–4085

Zak D, Reuter H, Augustin J, Shatwell T, Barth M, Gelbrecht J, McInnes RJ (2015) Changes of the CO_2 and CH_4 production potential of rewetted fens in the perspective of temporal vegetation shifts. Biogeosciences 12:2455–2468. doi:10.5194/bg-12-2455-2015