Wetlands (2021) 41: 71
https://doi.org/10.1007/s13157-021-01465-y

PEATLANDS

Drainage Ditches Contribute Considerably to the CH_4 Budget of a Drained and a Rewetted Temperate Fen

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Received: 10 February 2021 / Accepted: 24 May 2021 / Published online: 21 June 2021

Abstract

Small water bodies including drainage ditches can be hotspots for methane (CH_4) emissions from peatlands. We assessed the CH_4 emissions of a drained and a rewetted temperate fen including emissions of managed and unmanaged drainage ditches over the course of 2.5 years, covering three vegetation periods. Ditch CH_4 emissions in the rewetted fen were significantly higher than in the drained fen. In the rewetted fen ditches contributed up to 91% of the annual CH_4 budget, despite covering only 1.5% of the area. In the drained fen CH_4 emissions were solely made up of ditch emissions. When including CH_4 uptake by the peat soil, the CH_4 balance of the drained fen was neutral. Dissolved organic carbon concentrations likely had an enhancing effect on CH_4 emissions while nitrate and sulfate in the ditch water seem to have had an inhibitory effect. Air and water temperature controlled seasonal variability of ebullitive as well as diffusive CH_4 emissions. Ebullition contributed less than 10% to the overall CH_4 budget in the ditches. Drainage ditches represent a hotspot of CH_4 emissions and need therefore be taken into account when assessing the success of rewetting projects of peatlands.

Keywords Methane (CH_4) · Peatland rewetting · Restoration · Ditches · Methane budget · Ebullition

Introduction

Peatlands are a globally important carbon store (Treat et al. 2019) that is turned into a strong source of greenhouse gases (GHGs) when drained and faces other threats, for instance, from global warming (Loisel et al. 2021). Peatland rewetting represents an efficient way to reduce or stop GHG emissions (Günther et al. 2020). Due to the high availability of organic substrate in the soil, water-logged areas in drained or rewetted peatlands can become hotspots for emissions of the GHG methane (CH_4) in particular if the peat soils become inundated (Zak et al. 2015; Hahn et al. 2015). It was shown before that under ongoing inundated conditions a detritus mud layer is formed that turned out to be a substrate favorable for methane production (Zak et al. 2018). However, only a relatively small number of studies have so far examined the importance of CH_4 emissions from drainage ditches in peatlands. Drainage ditches can be important hotspots for CH_4 emissions in wetlands (Schrier-Uijl et al. 2011; Schrier-Uijl et al. 2010; Teh et al. 2011), sometimes contributing a major part of the total regional CH_4 budget (Schrier-Uijl et al. 2010). The overall importance of CH_4 emissions from ditches has recently been shown since they contribute up to the equivalent of 3% of the total global anthropogenic CH_4 emission (Peacock et al. 2021). Accordingly, in agricultural landscapes drainage ditches may contribute significantly to the landscape carbon budget via high CH_4 emissions (Peacock et al. 2017). Further, drainage ditches are considered to be an important source of CH_4 in settings where drained organic soils prevail (Drösler et al. 2013; Vermaat et al. 2011) and regionally even outweigh a terrestrial CH_4 sink (Bastviken et al. 2011; Korkiakoski et al. 2017). In this context, ebullition is often mentioned as an important pathway of CH_4 emissions in various aquatic ecosystems (Baulch et al. 2011; Bastviken et al. 2004; Repo et al. 2007; Yang et al. 2020).

The major biotic factor driving high CH_4 emissions is thought to be the trophic state of the water body (Peacock et al. 2021; Schrier-Uijl et al. 2011). Phosphate (PO_4^{3-}) as an indicator for the trophic status and reduced iron as an indicator for anaerobic conditions can explain a large proportion of the variance in CH_4 emissions (Schrier-Uijl et al. 2011). In connection with the trophic status of water bodies, the oxygen

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concentration in the water column is a good indicator for CH_4 emissions (Liikanen and Martikainen 2003). Since methanogenesis depends on small organic carbon molecules, either carbon dioxide (CO_2), hydrogen (H_2) or acetate (CH_3COOH) as a substrate (Kelly and Chynoweth 1981), the concentrations of dissolved organic matter (DOM) or dissolved organic carbon (DOC) are important drivers of CH_4 emissions in small water bodies (Bastviken et al. 2004; Zhou et al. 2019). However, it is often unclear whether the organic matter in ditches mainly derives from high biomass production within the ditch or from allochthonous DOC that was potentially leached at high rates from surrounding decomposing peat as was shown in a mesocosm experiment (Laine et al. 2014; Evans et al. 2015). Allochthonous organic material, however, is also known to be a less favorable substrate for methanogenesis (Zak et al. 2018) in comparison to dying plant material in unmanaged ditches (Zak et al. 2015).

High nutrient inputs from surrounding agriculturally-used peat soils can cause eutrophication in the ditches and thereby enhance plant and algal biomass production and subsequent depletion of oxygen from biomass decomposition (Zhou et al. 2019). Increased nutrient concentrations and experimental warming showed an increase in CH_4 emissions from small water bodies in a study on CH_4 ebullition from lake mesocosms (Davidson et al. 2018). This relationship was also shown in natural northern lakes and ponds (DelSontro et al. 2016). As was found in a meta-study, generally, shallow water bodies such as ponds are highly susceptible to warming and eutrophication because of their small water volume. Climate warming is expected to globally increase the CH_4 emissions via ebullition by up to 6–20% (Aben et al. 2017). Hence, CH_4 emissions from drainage ditches can be considered to further increase in global importance.

Here, we study the importance of ditch CH_4 emissions in GHG budgets on a landscape scale and assess the drivers for temporal variation in two peatlands with differing land use. We determine the effects of climatic (air temperature, water temperature, air pressure), biotic (DOC, nutrients) and morphological (water depth, orientation) variables on CH_4 emissions from ditches and evaluate the importance of ebullitive CH_4 fluxes in relation to diffusive fluxes. Using a 2.5 year time series of floating chamber measurements and closed chamber measurements in the adjacent peatland, we assess the interannual variability of CH_4 fluxes and seasonal CH_4 budgets.

Materials and Methods

Site Description

The two studied fens are located 8 km apart in the valleys of the two rivers Recknitz (drained fen) and Trebel (rewetted fen) in north-eastern Germany. The average annual mean temperature is 9.1 °C (DWD raster data, Krähenmann et al. 2016, reference period 1981–2010). The drained fen (PD, 54.13194° N, 12.62889° E, elevation a.s.l = 20 m) is an extensively used grassland that is harvested once a year for fodder production. The rewetted fen (PW, 54.10111° N, 12.73944° E, elevation a.s.l. = 2 m) has been rewetted since 1997 after having been used as an intensive grassland for decades. Rewetting in PW was accomplished by ditch-blocking. After rewetting, the water table in PW now fluctuates around the soil surface which was also the case during an earlier study at this site (Günther et al. 2015). Peat thickness is around 5 m in PD and approx. 6 m in PW. The peat in both sites is mainly of sedge and reed origin (Jurasinski et al. 2020). The vegetation at PD can be characterised as a uniform grassland dominated by *Ranunculus repens* L. and *Deschampsia cespitosa* (L.) P. Beauv. PW is dominated by sedges (*Carex acutiformis* Ehrh.) and occasional great willowherb (*Epilobium hirsutum* L.) and grey sallow (*Salix cinerea* L.). Especially around ditches and former peat cuttings large areas of common reed (*Phragmites australis* Trin. ex Steud.) and occasional common cattail (*Typha latifolia* L.) can be found.

Study Setup

At each study site (PD and PW) a soil measurement site was established inside a fenced area (12 × 30 m) between April and June 2017 (Fig. 1). Five collars for the measurement of soil CH_4 exchange were installed at a depth of 10 cm along a boardwalk at both sites. The soil collars also included vegetation that was representative of the site. Weather stations inside both fenced areas recorded air temperature, humidity, photosynthetic photon flux density (PPFD), wind speed, wind direction and precipitation (logged with CR300, Campbell Scientific, Bremen, Germany). Additionally, air pressure, vapor pressure and sunshine duration was obtained from three different weather stations in proximity of the soil sampling site (Warnemünde - 40 km NW, Barth - 30 km N, Greifswald, 40 km E), run by the German weather service (DWD). For analyses, the values of all three weather stations were averaged. Additionally, soil temperature was recorded automatically (HOBO Pendant, Onset, Bourne, USA) at three spots in two different depths (5 and 15 cm) along the board walk at the soil measurement site.

We selected two ditches to measure diffusive and ebullitive CH_4 exchange from the water surface in close proximity to each soil sampling site (~300–400 m distance, Fig. 1). At each site, one of the selected ditches runs parallel to the drainage direction (PD-p, PW-p) and one ditch runs orthogonal to the drainage direction (PD-o, PW-o) towards the main river. The differing orientations were chosen since we hypothesized that parallel ditches would show higher water flow velocities than parallel ones, potentially influencing water biogeochemistry. In all four ditches five sampling spots were established at approximately 10 m from each other (20 ditch sampling locations in total, Fig. 1).

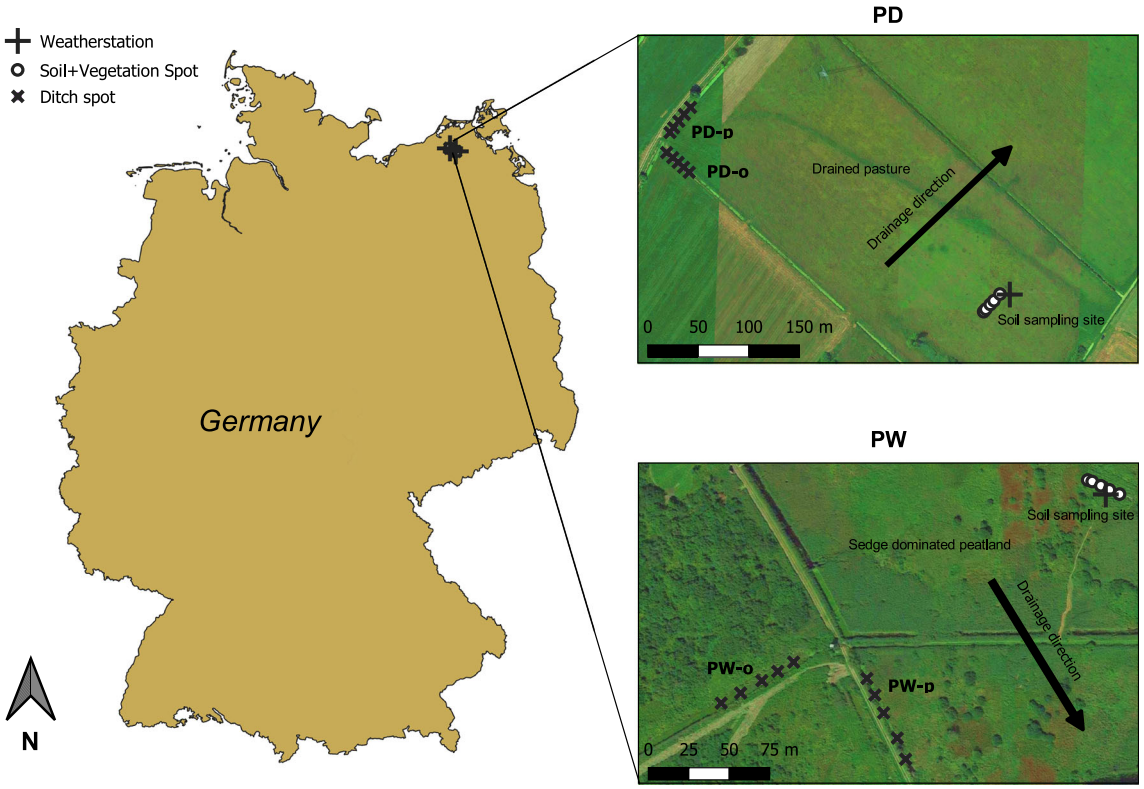

Fig. 1 Location of the study area in Germany and measurement locations for soil and ditch emissions at the drained (PD) and the rewetted fen (PW). Arial imagery by google maps ©

The ditches at PD are relatively uniform with a width of approximately 2 m and are regularly excavated in summer (own observations). Accordingly, the water depth of the ditches varies throughout the year, ranging from 10 to 70 cm. During summer, the ditches are often covered by common duckweed (*Lemna minor* L.) Further, water starwort (*Callitriche palustris* L.) was abundant. At PW the ditches are not managed and, thus, do not vary in depth over the year. PW-o, however, is significantly deeper than PW-d with average water depths of 104 cm and 38 cm, respectively. Also, PW-o is much wider than PW-d with approximately 4 m and 2 m, respectively. The ditches at PW are often covered entirely with vegetation during the summer months, with *Stratiotes aloides* L. being dominant in PW-o and *T. latifolia* L. and *L. minor* L. being dominant in PW-p. The banks of both ditches in PW are dominated by *Ph. australis* (Cav.) Trin. ex Steud. Ditches at both sites showed a thick mud detritus layer on top of more solid ground. However, most of the detritus layer was excavated at PD each summer. Aside from the excavation of the sediment there was no other active management of the ditches.

Flux Measurements

Diffusive CH_4 Fluxes

Diffusive emissions of CH_4 from the ditches were measured with a floating chamber. The floating chamber was constructed using a bucket (diameter = 20 cm, height = 25 cm), coated with reflective material to reduce heating inside the chamber (Fig. S1, supplementary information). The chamber was equipped with a temperature and humidity sensor as well as with a fan powered by a 9 V battery mounted inside the chamber lid. The chamber was placed inside a float (square 50 × 40 cm, Styrodur, BASF, Ludwigshafen am Rhein, Germany) and connected to a 1.5 m long handle. The floating chamber was not anchored and thus could float in a radius of approximately 1 m. Methane concentration measurements were carried out in-situ with laser spectrometers ('Ultra-Portable Greenhouse Gas Analyzer', Los Gatos Research, Mountain View, USA and 'GasScouter', Picarro, Santa Clara, USA) connected to the chamber with flexible polyurethane tubes (inner diameter: 4 mm). Measurements lasted 180 s.

Diffusive CH_4 flux measurements on the soil surface at PD and PW were carried out with circular flexible chambers constructed out of polyurethane walls varying in height between 0.9 and 1.4 m, following Günther et al. (2014). The diameter of the soil chamber was 0.65 m. The soil chamber was also equipped with three fans at the chamber top ensuring constant mixing of the air inside the chamber. Diffusive fluxes from the ditches and the soil surface at both sites were measured fortnightly between April 1st 2018 and September 29th 2020.

Diffusive fluxes were estimated using the *fluxx* function of the package *flux* (Jurasinski et al. 2014) for R (R development

core team 2020). The slope between all concentration points was calculated and the median slope was used for flux estimation (median-based regression, Siegel 1982). All diffusive flux measurements were visually checked for signs of ebullition (i.e. strong, sudden increase in CH_4 concentrations). If an ebullition event was identified during a diffusive flux measurement, it was excluded from the calculation of annual CH_4 balances (157 fluxes excluded, 302 fluxes remaining at ditches in PD and 182 fluxes excluded, 374 fluxes remaining at ditches in PW). This procedure was carried with soil flux measurements accordingly. Here, 300 of 308 flux measurements were kept in PD and 246 of 306 in PW.

Ebullitive CH_4 Fluxes

Ebullitive CH_4 emissions were assessed during the vegetation period of 2018, between May 1st and September 20th. Bubble traps were installed floating in the middle of the ditches (five measurement points at each ditch). The bubble traps were constructed from inverted polypropylene funnels (15 cm diameter opening) connected to a 120 ml syringe that functioned as the gas reservoir, similar to the approaches of Molongoski and Klug (1980) and Baulch et al. (2011). The funnel and the syringe were attached to each other with an insoluble adhesive sealant and a three-way stop cock allowed sampling at the top of the trap (Fig. S2, supplementary information). To prevent large water insects such as water scavenger beetles (Hydrophilidae) from entering the bubble trap we covered the opening of the funnel with a net (polyvinyl chloride, net width 5 mm). The traps were provided with a 20 × 30 × 5 cm cuboid float (Styrodur, BASF, Ludwigshafen am Rhein, Germany). The bubble traps were fixed in place by cables running between the float and both banks of the ditch to prevent any disturbance to the sediment.

To prepare for gas collection all bubble traps were filled with water completely. During the time in which the trap is deployed, rising bubbles are trapped in the funnel and replace the water inside the trap. After approximately two weeks (11–14 days) the volume of the accumulated gas in the trap was noted by reading the printed scales on the syringes. Gas samples were taken from the headspace collected in the syringe without disturbing the bubble trap by laying a portable aluminium footbridge across the ditch. Because PW-o was too wide to reach both banks, the bridge was instead placed onto a small, permanently-installed wooden platform inside the ditch. Gas samples were taken with a 60 ml syringe and immediately transferred to 12 ml exetainers (Labco, Lampeter, UK). The final sample volume was approximately 35 ml, thus the sample was stored with overpressure. After sampling we refilled the bubble trap completely with water.

Due to the long deployment times, CH_4 concentrations of air caught inside the bubble traps may have decreased due to equilibration with the water column or CH_4 oxidation (McGinnis et al. 2006). To quantify this potential error, fresh bubbles were collected using a mobile bubble trap by intentionally disturbing the sediment to induce ebullition at random locations within the ditch. Fresh bubbles were always collected after the permanent bubble traps had been sampled.

Gas analyses were performed within one week using a gas chromatograph (Shimadzu GC, Kyoto, Japan) with a flame ionization detector. As concentrations of CH_4 varied strongly, the samples had to be diluted up to a factor of 1000 and measured in different sensitivity ranges of the gas chromatograph.

Final ebullition fluxes were calculated as such: We assumed that bubbles caught in the traps originated from an area of sediment that corresponded to the area of the funnel opening (~0.0176 m^2). Thus, we normalized the recorded gas volumes in the bubble traps to 1 m^2 and divided by the number of days since the last sampling (ml m^{-2} d^{-1}). Then, we multiplied this value with the CH_4 concentration measured inside the gas samples (ppb). When there was no gas sample taken (every two weeks alternating in between the ditches) the arithmetic mean of the CH_4 concentration from all gas samples was taken to estimate the ebullitive CH_4 emission. Every bubble was sampled for CH_4 concentration every four weeks, meaning that every second week the arithmetic mean was taken as an estimate for the CH_4 concentration.

The final ebullition flux $Flux_e$ was estimated by estimating the CH_4 bubble rate in moles according to eq. 1:

$$Flux_e = \frac{P \times V}{R \times T} \times c \times m \quad (1)$$

with P the atmospheric pressure [hPa], V the volume of gas measured inside the bubble trap [ml], R the gas constant (R = 0.0821), T the temperature in the laboratory during analyses (298 K), c the concentration of CH_4 in the gas sampled (% by volume) and m the molar weight of CH_4 (16.04 g mol^{-1}).

Greenhouse Gas Budgets

We used a combination of bootstrap, jackknife and linear interpolation of the fluxes to calculate seasonal budgets (Günther et al. 2017). For each measurement day, one flux value per flux subset (ditch or soil) and each site (PD or PW) was randomly chosen. This was repeated 100 times to obtain 100 different flux time series. Then, the area-under-curve (auc.mc function from the R package *flux*, Jurasinski et al. 2014) was calculated 100 times for each flux time series each time leaving out one flux value, leading to a total of 10,000 different CH_4 balances. For the final CH_4 balances per site and flux subset we calculated the average and standard deviation of all balances. Using this procedure yields a more robust estimate of the seasonal CH_4 budgets as it is more sensible for temporal variation than the simple average of all

flux measurements or a simple linear interpolation. The CH_4 balances were calculated per season (i.e. vegetation period (April — September) and non-vegetation period (October — March).

To estimate the contribution of CH_4 emitted from ditches to total ecosystem emissions we manually determined the area covered by ditches by digitizing them within a randomly-chosen 1 km^2 area around the soil sampling site using aerial imagery. The area share of ditches was approximately 1.52% and 1.49% in PD and PW, respectively. To derive the total contribution of ditch CH_4 emissions to the overall ecosystem CH_4 budget, the ditch budgets were weighed using their relative spatial share within the 1 km^2 area. The rest of the area was assumed to emit on average as much CH_4 as the soil sample locations.

Water and Sediment Characteristics

Water samples of ditch water and groundwater at the soil sampling site were taken to assess potential influences of chemical properties (i.e. nutrients, DOC) on CH_4 emissions. The approximate water depth of the ditches was measured with a measuring stick for every measurement spot at the approximate center of the ditch every time flux measurements were done. Additionally, water flow velocity was measured at irregular intervals with a water flow meter (OTT MR pro, OTT, Kempten, Germany). Ditch water samplings took place on eight irregular occasions, however, covering all seasons between April 2018 and March 2019. On each sampling occasion, one sample was taken for every flux measurement location in the ditch ($n = 20$). Groundwater samples were taken at the central site every four weeks between April 2018 and September 2020. The groundwater samples were obtained from three water gauges per site which were located close to the soil surface measurement plots. Samples were obtained with a tube connected to a float, ensuring that water samples were always taken at the same depth (5 cm). All water samples were directly filtered in the field with syringe filter units (pore size 0.45 μm, Sartorius, Göttingen, Germany) and afterwards stored cool (~5 °C) or frozen until analysis. The water samples were analyzed for DOC, dissolved inorganic carbon (DIC), phosphate (PO_4^{3-}), nitrate (NO_3^-), ammonium (NH_4^+) and total nitrogen (TN). Dissolved organic carbon (DOC), DIC and TN were analyzed using a Dimatoc 2100 (Dimatec, Essen, Germany). All other nutrients (PO_4^{3-}, NO_3^-, NH_4^+ and NO_2^-) were analyzed with an AA3 SEAL Auto Analyzer 3HR continuous flow analyzer (SEAL Analytical, Norderstedt, Germany). Sulfate (SO_4^{2-}), chloride (Cl^-), bromide(Br^-) and fluoride (F^-) with an ion-chromatograph (930 Compact IC Flex, Metrohm, Herisau, Switzerland). Further, a Multiprobe AP 2000 (Aquaread, Bridge House, UK) was used to measure pH, O_2 saturation [%], water temperature [°C], electrical conductivity [μS cm^{-1}], redox potential [mV] and salinity [μS] directly in the ditches at a depth of approximately 15 cm. Water levels at the soil sampling sites were measured continuously with an automatic logger located in the central of the three water gauges (BlueCon 2, SEBA Hydrometrie, Kaufbeuren, Germany.

In June 2018 sediment samples were taken in two depths (0–5 cm, 10–20 cm) from the sediment surface of each ditch. The samples were dried for 24 h at 105 °C and subsequently ground for three minutes. Carbon, nitrogen and sulfur concentrations in the sediment samples were analyzed on a vario EL cube CNS analyser of elementar (Hanau, Germany).

Statistical Analyses

All statistical analyses and visualizations were carried out with R 4.0.2 (R development core team 2020). The entire dataset and any subsets were tested for normality and homogeneity of variance using Shapiro-Wilk tests and Levene's tests, respectively. Where data was non-normally distributed or the variance was not homogeneous, Kruskal-Wallis tests were used to detect significant differences between subpopulations within the dataset. In order to assess relationships between water chemical variables (pH, O_2 saturation, water temperature, electrical conductivity, redox potential and salinity) and CH_4 fluxes water chemical variables and ditch CH_4 fluxes were averaged (mean) by date – only when water samples from the ditch water were taken - because not every flux measurement had an associated measurement of water chemical variables. These relationships were tested for each ditch seperately. Diffusive CH_4 fluxes were log-transformed in order to achieve a near normal distribution. Transformed diffusive flux values were subsequently directly linked to environmental variables from the weather station with multiple linear regressions (wind direction, wind speed, air temperature, air pressure, vapor pressure and pressure change over different time intervals). Flux values were chosen as the response variable with the environmental variables as potential explanatory variables. Variables were chosen stepwise by backward selection. Additional variables, such as concentration values of nutrients and DOC that were either only available at certain dates were merged with daily average ebullitive and average non-transformed diffusive CH_4 fluxes and analyzed separately with the same approach (multiple linear regressions with stepwise backward selection).

Results

Environmental Variables and Ditch Characteristics

According to the weather station data air temperature differed only slightly between PD and PW. However, during the two-year study period PD was much drier than PW, receiving only

973 mm in comparison to 1173 mm at PW. Nonetheless, there were pronounced dry spells in both peatlands during the summer months of 2018 and 2019 (Fig. 2).

In PD water levels remained close to the soil surface in winter and were very low in summer (overall mean = −28 cm, overall minimum = −78 cm) whereas in PW water levels were more stable (overall mean = −0.5 cm, overall minimum = −28 cm) (Fig. 2). Soil temperatures were higher at PD than at PW, reflecting the overall drier conditions (10.3 °C at PD and 9.1 °C at PW). Due to extremely dry conditions in the summers of 2018 and 2019, the water levels in the ditches varied strongly over the seasons. Ditches at PD repeatedly fell dry in late summer. Generally, the amplitude of water table fluctuations in the ditches was lower in PW.

Nutrient concentrations in the ditches of PD and PW differed strongly. The two ditches at PD showed significantly higher concentrations of nitrate (Table 3, $\chi^2 = 39.95$, d.f.: 1, $p < 0.01$). DOC concentrations in the ditch water were significantly higher in PW compared to PD (Table 3, $\chi^2 = 38.78$, d.f.: 1, $p < 0.01$). Regardless of the site, ditches with an orthogonal orientation to the drainage direction (PW-o and PD-o) showed DOC concentrations that were almost double those of the ditches with parallel orientation to drainage direction (Table 3). Strikingly high sulfate concentrations were found in all ditches, especially PD-p and PW-p. Here, the parallel ditches showed similar and high concentrations, where the orthogonal ditches showed lower concentrations. (Table 3).

Also, the sediment samples taken in the orthogonal ditches (PD-o, PW-o) showed much higher concentrations of carbon, nitrogen and sulfur (Table 1). In PW the concentrations of phosphate differed significantly between the ditches ($\chi^2 = 13.35$, DF: 1, $p < 0.01$) with PW-o having higher values than PW-p. DOC concentrations also differed significantly across sites ($\chi^2 = 33.05$, DF: 1, $p < 0.01$).Concentrations of DOC in the groundwater differed significantly between the two sites ($\chi^2 = 131.26$, d.f.: 1, $p < 0.01$). Average groundwater DOC concentrations at PD were more than four times higher than at PW (96.6 ± 44.1 vs. 20.7 ± 19.5 mg l^{-1}). Concentration values of other nutrients in the groundwater can be found in supplementary information (Table T1).

Diffusive Fluxes

All ditches were strong sources of CH_4 during the measurement period. Diffusive CH_4 fluxes from ditches were generally significantly higher than soil fluxes from the adjacent peatlands. This holds true for both PD and PW (PD: $\chi^2 = 358.59$, d.f.: 1, $p < 0.01$; PW: $\chi^2 = 259.66$, d.f.: 1, $p < 0.01$). Maximum CH_4 fluxes from the ditches were reached during summer months with up to 1469.5 mg m^{-2} h^{-1} for PW and 464.7 mg m^{-2} h^{-1} for PD (Fig. 3). The average diffusive CH_4 fluxes from the ditches were higher in summer than in winter by up to three orders of magnitude and they differed strongly between different ditches (Fig. 3). Ditches at PW showed signifcantly higher diffusive CH_4 fluxes than at PD (75.7 ± 213.3 mg m^{-2} h^{-1} vs. 13.8 ± 37.4 mg m^{-2} h^{-1}, $\chi^2 = 52.19$, d.f.: 1, $p < 0.01$, Fig. 3). Also, within one site there were large differences between ditches. Orthogonal ditches (PD-o and PW-o) emitted much more CH_4 than parallel ones (PD-p and PW-p) in both sites ($\chi^2 = 181$, d.f.: 1, $p < 0.01$, Fig. 4a). CH_4 fluxes from soils ranged around or slightly below 0 in PD, while soils in PW were on average a weak source of CH_4 (0.5 ± 1.6 mg m^{-2} h^{-1}) with a maximum CH_4 flux (15.4 mg m^{-2} h^{-1}) recorded on May 3rd 2018.

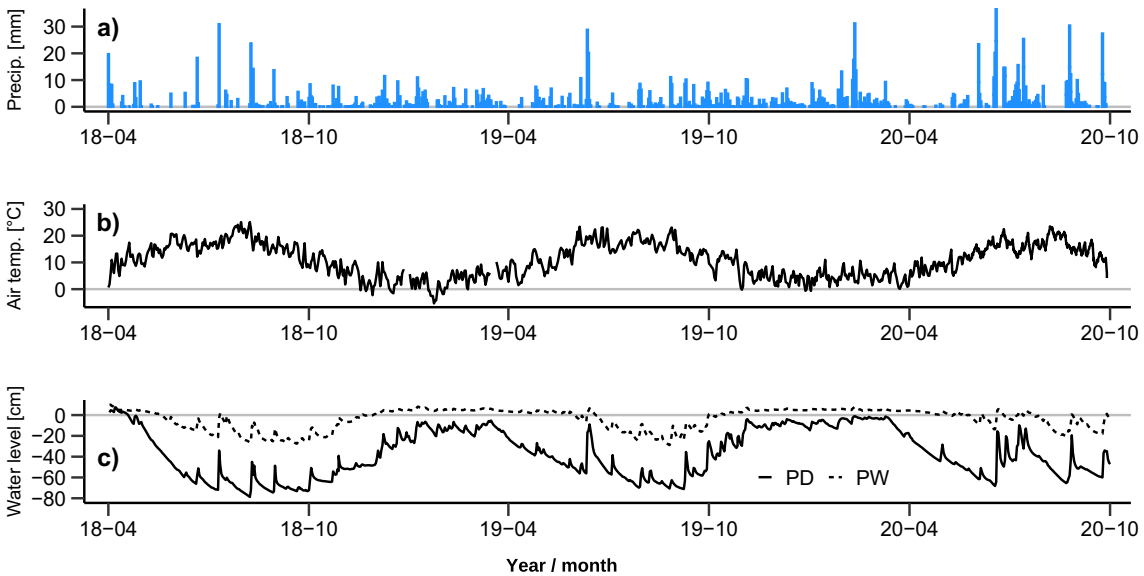

Fig. 2 Seasonal course of **a)** daily precipitation, **b)** daily mean air temperature and **c)** groundwater level at PD and PW. At c) the solid line depicts PD and the dashed line depicts PW. Daily precipitation and daily mean air temperature were averaged between PD and PW, since differences were not observable due to the low distance of the sites

Table 1 Nutrient contents of carbon (C), nitrogen (N) and sulfur (S) [% dry weight] of sediment samples in ditches (± denotes one standard deviation, $n = 4$ per ditch). PD denotes the drained site, PW, the rewetted site. **p** denotes ditches parallel and **o** ditches orthogonal to drainage direction.

	PD-p	PD-o	PW-p	PW-o
C	9.7±4.9	29±1.7	7.7±3.2	38±2.4
N	0.7±0.4	1.9±0.1	0.6±0.3	2.5±0.1
S	0.5±0.4	3.8±0.2	0.3±0.1	1.6±0.2

Ebullitive Fluxes

Ebullitive fluxes showed a seasonal pattern with highest fluxes between July and September (maximum CH_4 flux: 23.4 mg m^{-2} h^{-1} at PW-p on August 21st 2018) (Fig. 5). The average ebullitive CH_4 flux was significantly higher at PW than at PD (7.0 ± 4.5 mg m^{-2} h^{-1} vs. 2.7 ± 2.7 mg m^{-2} h^{-1}, $\chi^2 = 64.15$, d.f.: 1, $p < 0.01$). However, there were no significant differences between the two ditches within one site (PD: $\chi^2 = 2.57$, d.f.: 1, $p = 0.11$; PW: $\chi^2 = 1.01$, d.f.: 1, $p = 0.31$; Fig. 4b). The bubble rate, estimated with the bubble traps differed significantly only between PW-p and PW-o ($\chi^2 = 7.83$, d.f.: 1, $p < 0.01$). However, CH_4 concentrations in the gas samples taken from the bubble traps were significantly higher in the ditches at PW than at PD ($\chi^2 = 86.37$, d.f.: 1, $p < 0.01$, Fig. 4b).

CH_4 Budgets

Seasonal CH_4 budgets were roughly ten times lower in winter than in summer. In both sites, highest seasonal CH_4 emissions were estimated for ditches in summer 2018 (Table 4). Non-ditch CH_4 emissions in PD were negligible while PW was a weak source. In summer 2018 approximately 9.1% and 2.5% of the total ditch CH_4 emissions were transported via ebullition in PD and PW, CH_4 emissions differed strongly from year to year for both ditches and adjacent peatlands. Especially seasonal soil CH_4 emissions at PW decreased by approximately 90% when comparing summer 2018 and summer 2019. Also, the ditch CH_4 emissions declined by 68%, comparing summer 2018 and summer 2019 at PW while they were comparably stable at PD. Winter CH_4 emissions from the ditches roughly made up between 7.5 and 15% of the annual ditch CH_4 budgets in PD and PW, respectively.

Ditches in PD and PW covered only 1.52 and 1.49% of the area, respectively. Still, CH_4 emissions from ditches were of high relevance for the total ecosystem CH_4 budgets (Table 5). Because emissions from ditches greatly exceeded the weak sink or source from peat soils in PD and PW, ditches dominated the total ecosystem CH_4 budgets in both sites.

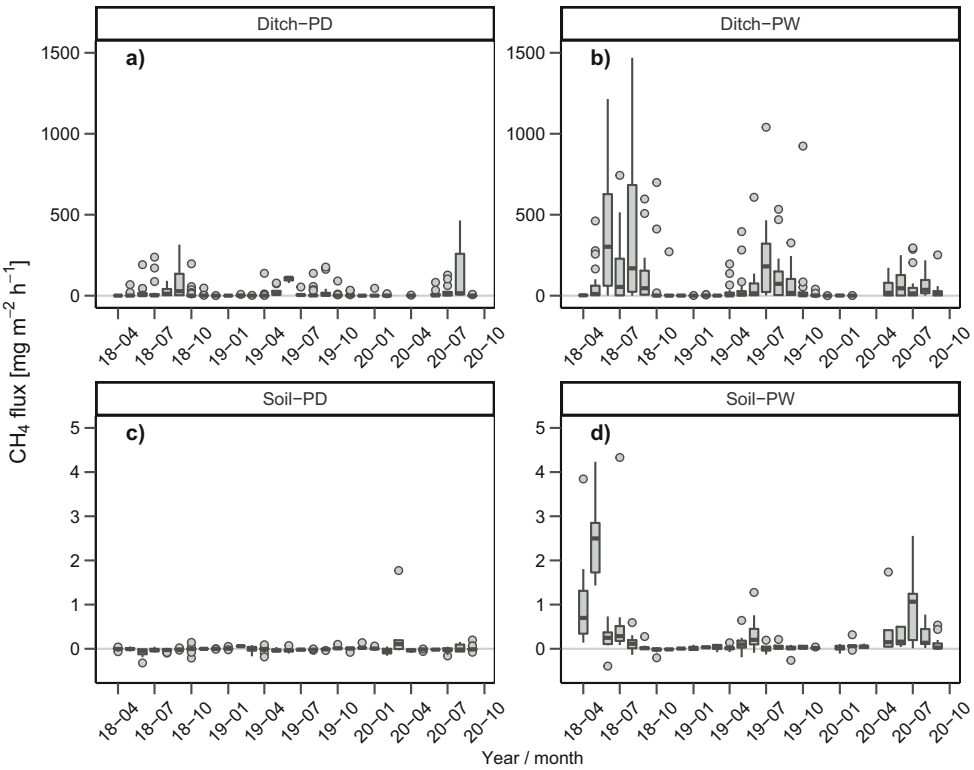

Fig. 3 Seasonal course of diffusive CH_4 fluxes from ditches at PD **(a)** and PW **(b)**, and soil surface at PD **(c)** and soil surface at PW **(d)**. Note the differing y-axes between upper and lower panels. Panel **(d)** does not show the single maximum diffusive flux from the soil surface

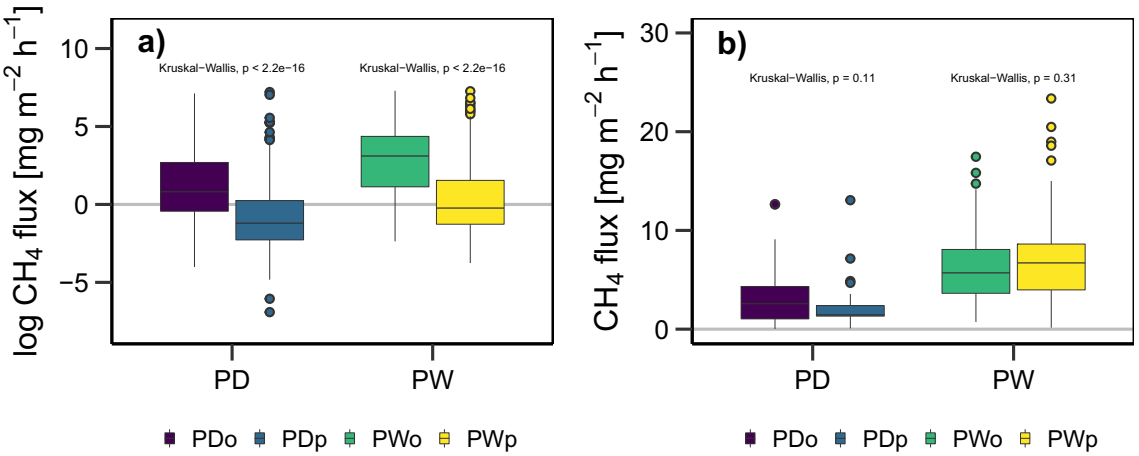

Fig. 4 Boxplot of **a)** log-transformed diffusive CH_4 fluxes and **b)** ebullitive CH_4 fluxes by ditch. Kruskal-Wallis tests were used to test for differences between the ditches

Drivers of Diffusive and Ebullitive CH_4 Emissions

Both approaches of linking the diffusive CH_4 fluxes to water chemical parameters or weather variables — using daily averages and direct flux values — revealed that air temperature was the most important factor for explaining seasonal variation at both sites. Looking only at climatic variables that were available for every diffusive flux measurement, air temperature and water depth were significantly positively correlated with the diffusive CH_4 flux at PW but still could only explain 14% of the overall variability ($R^2 = 0.14$, F = 24.45, DF: 3 and 422, $p < 0.01$). At PD air temperature and solar radiation were significantly positively correlated with diffusive CH_4 flux. However, the explanatory power of the multiple linear regression was very low ($R^2 = 0.09$, F = 12.52, DF: 3 and 378, $p < 0.01$). Considering all averaged diffusive CH_4 fluxes and additional water chemical variables as well as other daily climatic variables regardless of the site, the explanatory power of a multiple linear regression increased to 45% ($R^2 = 0.45$, F = 6.12, DF = 4 and 30, $p < 0.01$). Again, air temperature showed the strongest positive relationship with diffusive CH_4 fluxes. However, in this multiple linear regression across sites with

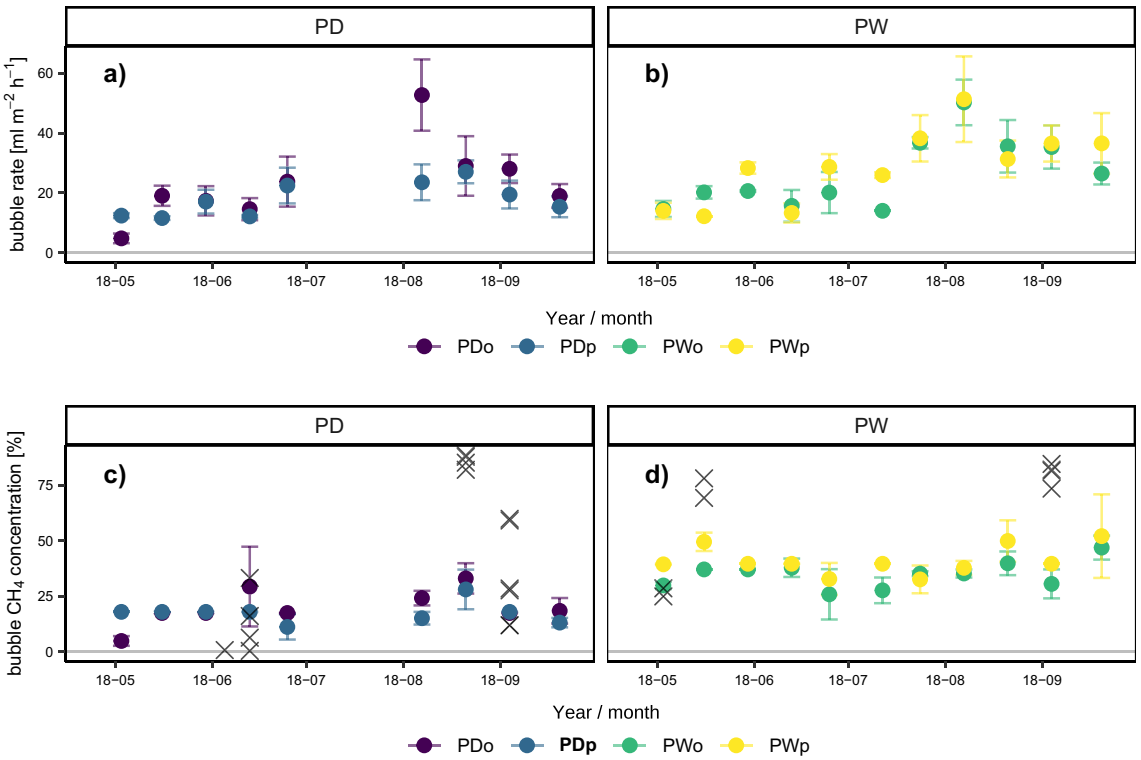

Fig. 5 Bubble rate recorded at the bubble traps at PD **(a)** and PW **(b)** and CH_4 concentrations in air samples from the bubble traps at PD **(c)** and PW **(d).** Crosses indicate the CH_4 concentrations of individual fresh bubble samples

averaged values, ditch water depth, wind speed and vapor pressure were also significantly positively correlated with diffusive ditch CH_4 fluxes. The relationships between the averaged explanatory variables and averaged CH_4 emissions held true for PW. At PD, using the averaged fluxes and explanatory variables, none of the variables were correlated with CH_4 emissions. Thus, it is likely that single high fluxes at PW dominated the multiple linear regression models. Summary statistics of the regression models can be seen in the supplementary information (Table T2).

Nutrient concentrations varied among ditches of different orientation at both sites (Table 2). The much higher nitrate concentrations in PD-p compared to PD-o reveal a pattern that is consistent with significantly lower diffusive CH_4 fluxes in PD-p than in PD-o ($\chi^2 = 82.19$, DF: 1, $p < 0.01$). In connection with this, the pattern of higher DOC concentrations in the ditch water at PW (Table 1) may stand in connection to overall higher diffusive CH_4 fluxes at PW. Thus, it is likely that ditches with higher DOC concentrations consistently exhibited higher diffusive CH_4 fluxes and that ditches with higher nitrate concentrations potentially emit lower quantities of CH_4. This is the only apparent connection between nutrient concentrations and diffusive CH_4 fluxes. Linear regressions between averaged nutrient concentrations (including DOC) by ditch failed to produce significant relationships (supplementary information, Table T2). Daily diffusive CH_4 fluxes averaged across all ditches exhibited a moderate positive relation with vapor pressure (Fig. 6).

Daily ebullitive CH_4 fluxes were significantly positively correlated with air temperature, wind speed and wind direction ($R^2 = 0.51$, $F = 8.29$, DF: 4 and 32, $p < 0.01$). When averaged by day and across both sites and all four ditches, ebullitive CH_4 fluxes exhibited a strong positive correlation with vapor pressure ($R^2 = 0.53$, $F = 10.15$, DF: 1 and 9, $p = 0.01$) (Fig. 6). Air pressure as such was not significantly correlated with ebullitive CH_4 flux. Correlations between all explanatory variables and ebullitive or diffusive CH_4 fluxes are shown in the supplementary information (SF1 and SF2).

Discussion

CH_4 Emissions from Ditches and Soils

CH_4 emissions from ditches were much higher than those from the peat soils in both the drained (PD) and the rewetted peatland (PW). This is in line with other studies who find ditches to be important sources of CH_4 emissions in peatlands (Drösler et al. 2013, Schrier-Uijl et al. 2011, Van Den Pol-Van Dasselaar et al. 1999). The much higher ditch CH_4 emissions at PW compared to PD highlight the importance of unmanaged drainage ditches also for the greenhouse gas balance of rewetted peatlands. Overall, the magnitude of the diffusive ditch CH_4 emissions in our study compare well to other studies, depending on whether the surrounding peatland was in a rather natural state (PW) or drained for agriculture (PD). Generally, emission rates and maxima of the diffusive fluxes in PW were similar to values reported in similar studies on diffusive CH_4 emissions from ditches in temperate fens that we are aware of (Peacock et al. 2017; Evans et al. 2015; Vermaat et al. 2011) and also in the range of flux values reported from northern drained forests (Roulet and Moore 1995), but higher than values from infilled and vegetated ditches in a blanket bog (Cooper et al. 2014). Furthermore, emission rates from PD compare well with Crawford and Stanley (2016) that studied small-stream CH_4 emissions in an agriculturally used landscape. Generally, seasonal variation of diffusive ditch CH_4 emissions in our study was very high with distinct maxima in July and August. Average emissions in summer were 5 to 10 times higher than average winter emissions for PD and PW. Yet, winter emissions were consistent and not negligible.

Ebullitive fluxes showed maxima at both sites in summer 2018, which is in line with the known temperature dependence of CH_4 ebullitions (Davidson et al. 2018; DelSontro et al. 2016; Wik et al. 2013). Average CH_4 ebullitive fluxes were much lower than values reported from temperate ponds with high nutrient concentrations (Yang et al. 2020) and small lakes and ponds in the boreal region (DelSontro et al. 2016). Also, CH_4 concentrations in the air samples from the bubble traps were much lower compared to studies from temperate or

Table 2 Average water chemical properties of the ditches retrieved with the multiprobe. (± denotes one standard deviation, $n = 26$ per ditch) PD denotes the drained site, PW, the rewetted site. **p** denotes ditches parallel and **o** ditches orthogonal to drainage direction.

	PD-p	PD-o	PW-p	PW-o
pH	7.1±0.2	6.9±0.2	7.1±0.7	7.3±0.5
electrical conductivity [μS cm^{-1}]	860±183	719±250	669±376	729±288
Redox potential [mV]	−33±86	−62±69	−53±97	−53±110
Salinity [PSU]	0.4±0.1	0.3±0.1	176.7±306.1	65.7±178.6
O_2 saturation [%]	41±37	25±19	32±20	30±28

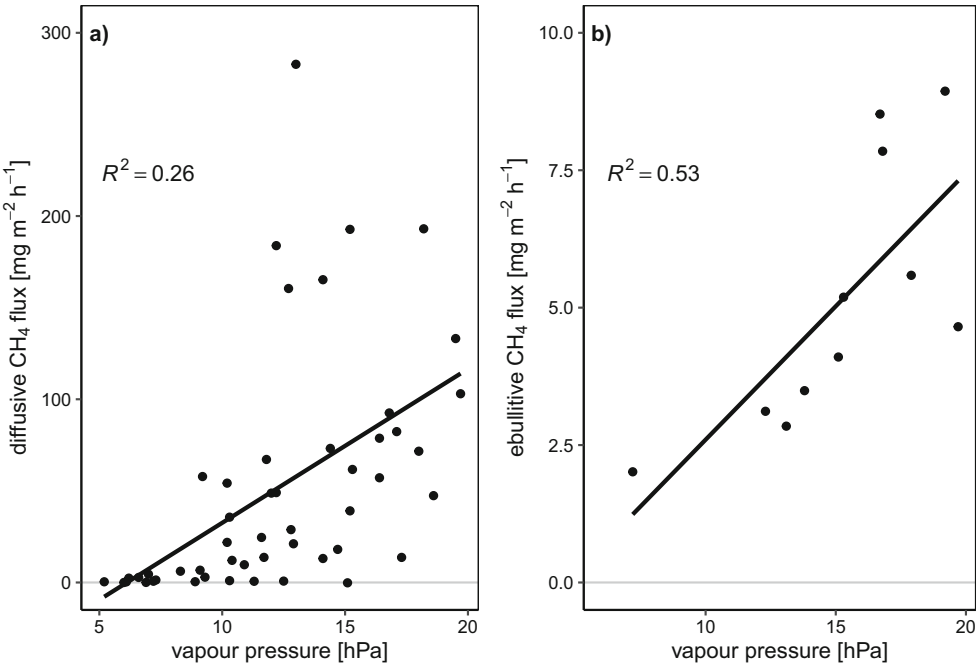

Fig. 6 Linear regression between vapor pressure and **a)** daily averaged diffusive CH_4 fluxes and **b)** daily averaged ebullitive CH_4 fluxes. Vapor pressure data was obtained from three weather stations of the German weather service (DWD) located within 40 km to the NW, N and E. vapor pressure values from all weather stations were averaged

subtropical regions (Maeck et al. 2013; Martinez and Anderson 2013). Instead, CH_4 concentrations in the bubbles and the associated ebullitive fluxes were rather comparable with values from subarctic peatlands (Wik et al. 2013). However, we have to stress that the sampling interval of the bubble traps may have led to significant re-oxidation of the CH_4 that accumulated in the bubble traps. Further, ebullition fluxes may be predominantly fuelled by autochthonous production while diffusive CH_4 flux may derive from allochthonous methanogenesis in the surrounding soils (Alshboul et al. 2016). This could potentially explain the relatively small contribution of ebullition to the total budget.

Unlike the PD ditches with consistent CH_4 emissions, PD soil was a small CH_4 sink during summer and a small CH_4 source during winter 2019/2020 which compares well with comparable studies (Van den Pol-van Dasselaar et al. 1998; Nykänen et al. 1995). The soils at PW showed lower average fluxes than found by a previous study at this site (0.6 mg m^{-2} h^{-1}, our study; 5–20 mg m^{-2} h^{-1}, Huth et al. 2013) and also lower than average values reported for temperate peatlands (4.5 mg m^{-2} h^{-1}, Turetsky et al. 2014). It is possible that methanogenesis in the soils was reduced in our study due to the drought conditions in the summers of 2018 and 2019 (Jurasinski et al. 2020).

Drivers of Ditch CH_4 Emissions

Diffusion as well as ebullition fluxes were explained best by temperature, as was also found in other small water bodies recently (Audet et al. 2020). Increased temperatures within aquatic environments are often associated with an increase of CH_4 production (Kelly and Chynoweth 1981; Duc et al. 2010). Thus, higher temperatures may have caused the increased CH_4 emission rates in summer by enhancing the metabolic rate of microorganisms most likely in the sediment of the ditches.

Other variables that have been found to influence CH_4 emissions of small water bodies are water depth (Vermaat et al. 2011; West et al. 2016), pH (Ye et al. 2012), or trophic status of the water body (e.g., Peacock et al. 2019). In line with this finding, the deepest ditch in our study showed the highest diffusive and ebullitive fluxes. However, given that the deepest ditch had the highest DOC and lowest sulfate concentrations it seems more likely that the DOCas a substrate or the lack of sulfate as an inhibitory factor was a more important driver of the CH_4 emission patterns. Further, we could not find a relationship between pH and CH_4 emissions. The range of pH in the ditch water in our study was relatively large (5–8). However, the ditches did not differ in their average pH values ranging at around 7 (Table 2). Chlorophyll A content seems to also be a good proxy for eutrophication and, thus, for CH_4 production and/or emission (e.g. West et al. 2016; DelSontro et al. 2018; Beaulieu et al. 2019). In our study, nitrate, phosphate and DOC contents can be seen as indicators for the trophic status of the ditches. Based on the analyses of the nutrient concentrations in the ditch water, especially the ditches at PD can be defined as eutrophic while the ditches at PW are rather mesotrophic. However, we did not directly determine the trophic status of the ditches. Regardless, we find ditches at PD to have much higher nutrient concentrations than ditches at PW. Regression models between averaged CH_4 emissions and nutrient/DOC contents failed to produce

significant results. However, individual ditches in our study differed substantially in their chemical properties, morphology and in their CH_4 emissions. Ditches with increased DOC and likewise lower sulfate concentrations (PD-o, PW-p, PW-o, Table 3) showed significantly higher diffusive and ebullitive CH_4 emissions, with PW-o standing out with uniquely high diffusive and ebullitive fluxes, DOC concentrations and low sulfate concentrations. This has also been shown in previous studies where TOC/DOC concentrations were related with CH_4 emissions in ponds and ditches (Crawford and Stanley 2016; Peacock et al. 2019).

Our analyses of nutrient concentrations in the ditches allowed for analyzing potential impacts of nutrient concentrations on CH_4 production and emission. For instance, nitrate is known to have an inhibitory effect on CH_4 production because it acts as a more favorable electron acceptor when other electron acceptors are limited(Watson and Nedwell 1998; Audet et al. 2020). The low CH_4 emissions in PD-p and PD-o may be explainable by the comparably high nitrate and/or high sulfate concentrations in these ditches. Diffusive CH_4 emissions from ditches in our study, however, were of similar magnitude in a recent study with comparable nitrate concentrations (Crawford and Stanley 2016). Sulfate is also known to inhibit methanogenesis (Lovley and Klug 1983; Dean et al. 2018; Zak et al. 2021). Ditches parallel to the general drainage direction (PD-p, PW-p) showed roughly double the sulfate concentration of the ditches running orthogonally to drainage direction. Sulfate concentrations in the investigated ditches and also in the porewater of the non-ditch measurements locations are comparably high. However, the closest official groundwater wells show consistently high sulfate concentrations and the sulfate concentrations in groundwater in the region are known to increase over time because the pyrite in the Pleistocene sediments is dissolved by the ever increasing nitrate concentrations in the percolation waters. And, depending on flow in the ditches, sulfate seems to accumulate Accordingly, these ditches showed lower diffusive and ebullitive CH_4 emissions. Hence, it is very likely that sulfate also plays a vital role within this system in inhibiting methanogenesis. However, considering the sulfate concentrations of the ditch water being so high and still showing CH_4 emissions is not necessarily contradictory as sulfate reduction may also occur very locally in micropores and therefore must not necessarily affect methanogenesis (Hahn et al. 2015).

Further, it is possible that due to a potentially lower water flow velocity and a longer residence time of the water in the orthogonal ditches, sulfate reduction can be more efficient and thus leads to lower concentrations and less inhibition of CH_4 production. However, lower water flow velocity could also lead to lower concentration of oxygen in the water, which is again favorable for methanogenesis. Finally, phosphorus is an indicator for eutrophication and, thus, for production, and therefore seems to be strongly related to increased methanogenesis and/or ebullition (DelSontro et al. 2016). Across both sites, the ditches with higher phosphate concentrations showed higher diffusive and ebullitive CH_4 emissions. Generally, it is important to note that the nutrient status can also indirectly influence CH_4 emissions through its effects on dominant vegetation (Davidson et al. 2015; Audet et al. 2020) Tables 4 and 5.

Other studies have found that water chemical properties are not always good predictors for CH_4 concentrations or fluxes in aquatic systems (Ortega et al. 2019). Apart from nutrient and substrate availability in the water column, the nutrient status of the sediment is important for methanogenesis. For instance, sediment accumulation rates are thought to be directly linked to the rate of methanogenesis (Maeck et al. 2013). Both PD-o and PW-o showed higher carbon contents in the sediment than PD-p and PW-p (Table 1). This difference may be driven by higher sediment accumulation rates from slower water flow velocities orthogonally to the general drainage direction. PD-o and PW-o also showed higher diffusive CH_4 emissions. Concluding, nitrate likely inhibited methanogenesis in PD ditches and higher DOC concentrations in PW-o may have led to higher CH_4 emissions. Here it is interesting to note that DOC concentrations in the groundwater were higher at PD than at PW (supplementary information, Table T1) which is characteristic for drained sites and indicates carbon leaching (Hyvönen et al. 2013). DOC concentrations at PD could further have been higher due to the drought conditions in the soil, potentially leading to higher percolation rates which in turns increases leaching. The pattern of increased DOC concentrations at the drained site is not reflected by the DOC

Table 3 Nutrient concentrations [mg l^{-1}] in the ditch water. (± denotes one standard deviation). DOC (dissolved organic carbon), DIC (dissolved inorganic carbon). PD denotes the drained site, PW, the rewetted site. **p** denotes ditches parallel and **o** ditches orthogonal to drainage direction.

	PD-p[a]	PD-o[b]	PW-p[c]	PW-o[d]
PO_4^{3-}	0.1±0.1	0.1±0.1	0.2±0.1	0.5±0.6
NH_4^+	0.1±0.1	0.4±0.6	0.2±0.6	0.2±0.4
NO_2^-	1.2±1.4	0.5±0.5	0.1±0.0	0.0±0.0
NO_3^-	21.3±12.0	6.8±11.3	0.3±1.00	0.5±1.3
SO_4^{2-}	120.0±11.6	79.8±27.2	123.7±48.2	41.3±33.5
Br^-	0.2±0.0	0.2±0.0	0.2±0.0	0.2±0.0
F^-	0.2±0.0	0.3±0.1	0.3±0.0	0.3±0.0
Cl^-	48.4±2.2	47.4±7.3	42.3±10.0	43.4±13.8
DOC	7.7±5.2	12.0±6.3	11.4±7.3	24.4±5.9
DIC	46.4±8.6	36.7±7.7	44.3±8.5	60.2±13.2

[a] for PO_4^{3-}, NH_4^+, NO_2^- and NO_3^- $n = 14$ | for SO_4^{2-}, Br^-, F^-, Cl^- $n = 9$ | for DOC and DIC $n = 46$[b] for PO_4^{3-}, NH_4^+, NO_2^- and NO_3^- $n = 12$ | for SO_4^{2-}, Br^-, F^-, Cl^- $n = 7$ | for DOC and DIC $n = 35$.

[c] for PO_4^{3-}, NH_4^+, NO_2^- and NO_3^- $n = 21$ | for SO_4^{2-}, Br^-, F^-, Cl^- n = 7 | for DOC and DIC $n = 47$.

[d] for PO_4^{3-}, NH_4^+, NO_2^- and NO_3^- $n = 20$ | for SO_4^{2-}, Br^-, F^-, Cl^- $n = 6$ | for DOC and DIC $n = 48$.

Table 4 CH_4 budgets in g m^{-2} per season for each flux subset (ditch and soil of both the drained (PD) and rewetted (PW) site). Summer season denotes the period between April 1st and September 30th and the winter season from October to March (± denotes 1 standard deviation).

Site	type	Apr. — Sept. 2018	Oct. — Mar. 2018/19	Apr. — Sept. 2019	Oct. — Mar. 2019/20	Apr. — Sept. 2020
PD	ditch diffusive	102.5±19	8.2±7.5	108.8±24	14.0±4.3	75.6±33
PD	ditch ebullition	10.3±1.5				
PD	soil diffusive	−0.1±0.02	0.0±0	−0.1±0.01	0.4±0.2	0±0
PW	ditch diffusive	919.6±147	65.1±17.0	293.5±75	29.6±14.5	123.2±28
PW	ditch ebullition	23.0±1.8				
PW	soil diffusive	4.0±0.7	0.04±0.0	0.4±0.1	0.11±0.0	1.3±0.3

concentrations in the ditch water (Table 3) where PW-p and PD-o have similar concentrations. This indicates that the low sulfate concentrations in PW-o may not lead to a suppression of methanogenesis as strong as in the other ditches (Table 3).

Since the residence time of organic matter is likely higher in the orthogonal ditches, the time that is available for matter decomposition and subsequent cycling is also increased. Generally, ditches in PW had higher macrophyte abundance which may also have provided an increased amount of organic matter as substrate for methanogenesis (Davidson et al. 2015).

Importance of Ditch Emissions for Total Ecosystem Budgets

The annual CH_4 budgets reported for PW-p and PW-o were much larger than values from other studies (Peacock et al. 2017; Schrier-Uijl et al. 2010). Annual budgets from PD-p and PD-o compared well to values reported from streams in agriculturally used landscapes (Crawford and Stanley 2016). However, when using simple linear interpolation to calculate seasonal budgets, values are much lower and comparing well with Peacock et al. (2017) (supplementary information T3). This, however is subject to a methodological debate. The approach used here to calculate seasonal budgets is more sensible to important short-lived peaks in emission patterns. Finally, we think that the relatively low flow of all ditches in PW and the accumulated, thick mud layer containing substantial amounts of easily degradable carbon, might be a reason for the high methane emissions at the wet site.

The relative importance of ditch emissions for total ecosystem CH_4 budgets seems very variable. Our values were far higher than values presented in a boreal peatland (Hyvönen et al. 2013) while they were very well comparable to a study in a temperate setting (Teh et al. 2011), where drainage ditches constituted a major hotspot for CH_4 emissions. At the rewetted peatland PW, relative contributions by ditches to ecosystem CH_4 budgets were still >50%.

Weather conditions were very dry during the study period, especially during the summer months. Thus, CH_4 emissions from the soils were likely lower than the long-term mean due to low water levels. Our results support the finding that ditches are hotspots of CH_4 emissions and, since extreme weather events are likely to become more frequent, could even gain importance in the future.

CH_4 emissions from ditches and soils continued during winter, although at a much lower rate. Overall, winter emissions contributed between 6 and 11% of the total annual ditch CH_4 emissions. When just looking at the winter season, CH_4 budgets

Table 5 CH_4 budgets [g m^{-2} $season^{-1}$] from ditches and soils weighed with their respective spatial share and the relative contributions [%] of the ditch emissions to the combined ecosystem CH_4 balance. Weighed seasonal balances of CH_4 emissions were multiplied with the spatial share of the respective landscape element (ditch, soil) estimated from an area of 1 km^2 around the soil sample locations. Summer denotes the period between April 1st and September 30th and winter the period from October to March.

PD	Apr. — Sept. 2018	Oct. — Mar. 2018/19	Apr. — Sept. 2019	Oct. — Mar. 2019/20	Apr. — Sept. 2020
Ditch diffusive	1.6±0.3	0.1±0.1	1.7±0.4	0.2±0.1	1.2±0.5
Ditch ebullition	0.2±0				
Soil diffusive	0±0	0±0	0±0	0.4±0.2	0±0
Combined ecosystem balance	1.8±0.3	0.1±0.1	1.7±0.4	0.6±0.3	1.2±0.5
Ditch contribution	100	100	92	33	100
PW	Apr. — Sept. 2018	Oct. — Mar. 2018/19	Apr. — Sept. 2019	Oct. — Mar. 2019/20	Apr. — Sept. 2020
Ditch diffusive	13.7±1.7	1.0±0.3	4.4±1.1	0.4±0.2	1.8±0.4
Ditch ebullition	3.5±0				
Soil diffusive	3.9±0.7	0±0	0.4±0.1	0.1±0	1.3±0.3
Combined ecosystem balance	21.1±2.4	1±0.3	4.8±1.2	0.5±0.2	3.1±0.7
Ditch contribution	82±12	100±30	92±25	80±50	58±22

consisted almost entirely of emissions from ditches in both sites. With data on this subject being so scarce, future studies should ideally involve measurements of winter emissions.

Contribution of Ebullition Fluxes

Ebullition contributed <10% to the overall CH_4 budgets of ditches in summer 2018. Many studies from various water bodies (shallow lakes, ponds, ditches, streams) report contributions of over 50% by ebullition to total CH_4 emissions (Tokida et al. 2007; Wilcock and Sorrell 2008; Baulch et al. 2011; Vermaat et al. 2011; Martinez-Cruz et al. 2017). However, few studies reported low importance of ebullition of between 10 and 38% (Minkkinen et al. 1997; Higgins et al. 2008). Moreover, CH_4 concentrations in the samples obtained from the bubble traps in our study were low compared to other studies (e.g. Maeck et al. 2013; Martinez and Anderson 2013), while fresh bubbles from the sediment showed higher CH_4 concentrations. This could be an indicator for measurement error due to relatively long residence times of the gas inside the traps prior to sampling and equilibration with the water in the trap. If the concentrations of the fresh bubbles only would have been used for the calculation of the ebullition, the contribution of ebullition to the overall CH_4 budget would have increased to 14% and 4% at PD and PW, respectively. Thus, ebullition would still be of minor importance.

Conclusions

Ditches can play an important role in the overall GHG budgets of peatlands. Here, we show that managed and unmanaged ditches in drained and rewetted temperate fens can act as hotspots for CH_4 emissions. Despite ditches covering only a small part of the peatlands, the total ecosystem CH_4 budget was periodically determined entirely by ditch emissions even in the rewetted fen. Emissions from ditches in the rewetted fen were much higher than from ditches at the drained site. High nitrate concentrations in ditches of the drained fen seemed to reduce CH_4 emissions. Further, high sulfate concentrations seemed to convergate with lower diffusive and ebullitive CH_4 fluxes at both the drained and the rewetted fen. Increased DOC concentrations in the ditch water seemed to foster CH_4 emissions at the rewetted fen. As unmanaged ditches tend to have higher macrophyte abundance, they can be particularly strong hotspots for CH_4 emissions. This must be considered in rewetting projects and filling these ditches must be taken into consideration, although data on the emissions of infilled ditches in temperate fens is lacking. In drained fens high CH_4 emissions from ditches add to the high CO_2 emissions from the drained peat soils. There, emissions from drainage ditches need to be included in the calculations of GHG budgets, in drained and rewetted peatlands alike.

Further, our results add to the evidence that in shallow water bodies high sulfate concentrations do not exclude high methane emissions because substrate for methanogenesis is very likely abundant and the formation of a thick mud layer may also foster the development of very steep process gradients with substantial methanogenesis likely going on slightly deeper within the mud whereas closer to the surface sulfate reduction may dominate. Investigating the layering of carbon processing in such systems might be quite insightful but appears challenging. Hopefully, future methodological advances will allow us to study these processes in high spatial and temporal resolution in situ.

Acknowledgements The authors would like to acknowledge the support of several student assistants and field technicians for their support in field measurements and laboratory analyses. We would also like to thank Jacob Mulder from the agricultural cooperative Schulenberg for letting us carry out our measurements on his property. Further,14-BM-A55-0030/16).

Availability of Data and Materials Data is available via PANGAEA link: https://issues.pangaea.de/browse/PDI-28277

Code Availability R code for analyses is available via PANGAEA link: https://issues.pangaea.de/browse/PDI-28277

Author Contribution DK, CW, AGand GJ contributed to the study design. DK an CW collected the data in the field and analyzed the data and did laboratory work. DK wrote the first version of the manuscript. AG, CW an GJ added comments which were incorporated by DK. All authors read and approved the final manuscript.

Funding Funding was provided by the European Social Fund in the WETSCAPES Project (grant number: ESF/14-BM-A55–0030/16)..

Declarations

Ethics Approval Not applicable.

Consent to Particip0061te Not applicable.

Consent for Publication Not applicable.

Competing Interests I Allajbeu: no conflict of interest to disclose
S Hickman: has research collaborations with Merantix, ScreenPoint, Volpara, and Lunit
N Payne: GE Healthcare funding for research
P L Moyle: no conflict of interest to disclose
K Taylor: no conflict of interest to disclose
N Sharma: GE Healthcare funding for research
F J Gilbert: GE Healthcare funding for research

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