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Temporally dynamic carbon dioxide and methane emission factors for rewetted peatlands

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Rewetting drained peatlands is recognized as a leading and effective natural solution to curb greenhouse gas emissions. However, rewetting creates novel ecosystems whose emission behaviors are not adequately captured by currently used emission factors. These emission factors are applied immediately after rewetting, thus do not reflect the temporal dynamics of greenhouse gas emissions during the period wherein there is a transition to a rewetted steady-state. Here, we provide long-term data showing a mismatch between actual emissions and default emission factors and revealing the temporal patterns of annual carbon dioxide and methane fluxes in a rewetted peatland site in northeastern Germany. We show that site-level annual emissions of carbon dioxide and methane approach the IPCC default emission factors and those suggested for the German national inventory report only between 13 to 16 years after rewetting. Over the entire study period, we observed a source-to-sink transition of annual carbon dioxide fluxes with a decreasing trend of -0.36 t CO_2-C ha^{-1} yr^{-1} and a decrease in annual methane emissions of -23.6 kg CH_4 ha^{-1} yr^{-1}. Our results indicate that emission factors should represent the temporally dynamic nature of peatlands post-rewetting and consider the effect of site characteristics to better estimate associated annual emissions.

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Peatlands are the most carbon-dense ecosystems of the terrestrial biosphere and effective carbon (C) sinks under anoxic conditions in their pristine state. In contrast, drained peatlands release C to the atmosphere as carbon dioxide (CO_2) due to the aerobic soil conditions in the peat layer. Drained peatlands only cover ~0.3% of the global land area yet are contributing disproportionately between ~3–5% of the total global anthropogenic emissions (in CO_2-eq)[1,2], a variance influenced by distinct historical period. Drained peatlands emit about 220 Mt CO_2-eq yr^{-1} in the EU (5% of total EU emissions)[1,3] and ~53.7 Mt CO_2-eq yr^{-1} in Germany (>7% of the total national greenhouse gas (GHG) emissions)[4]. The most effective method to halt CO_2 emissions from oxidation in drained peatlands is to raise the water level, re-establishing water-saturated conditions[5], hereafter referred to as rewetting. Rewetting helps to conserve peat C storage and to form new peat and is a viable long-term solution to re-establish a climate cooling trajectory. However, water level management plays a critical role for mitigation purposes, with the recommended optimum range of 0 to 0.2 m (i.e., close to the surface and slightly above the surface), which reflects the near-natural conditions[6–8] in terms of water table depth (WTD) conditions. Moreover, in temperate region, peatland C stocks are currently under multiple pressures due to climate change. These include heat waves and droughts, extreme precipitation and consequential hydrological changes, or nutrient loading as a consequence of flooding. Also, there are no studies estimating future climate effects on the mitigation potential from peatland rewetting[9]. While a substantial CO_2 emission reduction can be achieved by rewetting[10,11], raising the water table often leads to a sharp increase in methane (CH_4) emissions due to the established anaerobic conditions. As indicated in the Intergovernmental Panel on Climate Change (IPCC) 2013 Wetlands Supplement[5], rewetted peatlands emit considerably more CH_4 than undrained ones. This difference is considered to be on average 46% more when compared to the original pre-management (with varying conditions within the sites) emissions in northern latitude (40°−70° N) peatlands[12]. However, CH_4 has a short atmospheric lifetime of an estimated 12.4 years[13] and recent modeling efforts indicate that CH_4 radiative forcing does not undermine the climate change mitigation potential of peatland rewetting[14]. It has also been indicated that rewetting might not restore natural conditions of the peatlands immediately after rewetting or even within decades[15] but that rewetted peatlands may still cope better (compared to the drained peatlands) with the impact of extreme weather events, in terms of C storage and mitigation potential[5,16]. According to current estimates, rewetting prevents about 20–30 t CO_2-eq ha^{-1} yr^{-1}, depending on the subsequent land-use in European temperate regions[17]. However, the emission factors (EFs) underlying these estimates are mostly based on data that only represent a limited range of spatio-temporal variability in emissions, and thus require updating and refinement. Hence, more long-term national datasets will help with moving towards country-specific higher Tier EFs.

Natural climate solutions such as peatland rewetting are needed to reduce and avoid CO_2 emissions and eventually re-establish the CO_2 sink capacity[17–19]. In order to reach EU climate-neutrality by 2050, these natural solutions become increasingly relevant within the EU countries[17,20]. Germany, with approximately 95% of its organic soils drained[4,21] has set a goal of achieving C neutrality by 2045, proposed through the amendment to the Climate Change Act 2021. In Germany, peatlands—including bogs and fens—cover ~1.35 ± 0.07 Mha[22–25] (Fig. 1a indicates the updated numbers for organic soils in Germany) or 3.6 − 5% of the total area under land-use. However, peatland rewetting is currently not considered in the German national GHG inventory report (NIR) due to lack of comprehensive drainage status data[8] and presumably, lack of long-term data on C fluxes after rewetting. The IPCC default Tier 1 EFs and their applicable methodologies for rewetted organic soils have been used globally for both CO_2 and CH_4 emissions (via the guidance of Wetland Drainage and Rewetting under the Kyoto Protocol). However, due to large uncertainties, countries are encouraged to develop more detailed and dynamic EFs that fully capture the transient nature of C fluxes in the time since rewetting and at national level (Tier 2). These detailed/dynamic EFs would avoid assigning a sudden change to the steady-state emissions immediately after rewetting and reduce inaccuracy in estimating the true emission reduction potentials following rewetting. Few peer-review studies have published results of long-term annual C exchanges after rewetting, particularly temperate rewetted peatlands with modified water table regimes[2,10,26–28], using the eddy covariance (EC) technique. In view of more frequent extreme events, such long-term collection of in-situ measurements following rewetting is also needed to understand the impact of drought in temperate peatlands[26,29–31]. Here, we use in-situ EC measurements to evaluate the annual emissions after rewetting as well as the effect of the 2018 European summer drought on the emission trends for a rewetted site in Germany, exhibiting heterogeneity within the footprint of the EC tower (Fig. 1b). By using both time series of year-round flux measurements and high-resolution imagery used for land cover classification, we are able to assess the temporal changes in emissions caused by dynamic environmental conditions such as the vegetation development and water level fluctuations post-rewetting. We show how measured emissions compare to default estimates in a rewetted peatland transitioning from CO_2 source to sink and identify the main drivers of this transition.

Results and discussion

Temporal dynamics of emissions and their uncertainty. The ecosystem-scale annual balance at a northeastern German fen site changed dynamically for both CO_2 and CH_4 during the transitional period after rewetting. It took more than 13 years for our site to reach the EF_{CO2} and EF_{CH4} provided by the IPCC (Fig. 2 & Table 3.1 and 3.3 in IPCC 2014, 2013 Wetlands Supplement[5]) and those suggested for the German NIR (Fig. 2 and Table 2 in ref. [8]). It is not clear whether our site has reached a steady-state phase or continues to follow the observed trajectory. We found a negative trend for net ecosystem exchange (NEE) of CO_2 and CH_4 fluxes (Fig. 2a, b), which led to a CO_2-source-to-sink transition and a considerable reduction of the 100-year global warming potential (GWP, Fig. 2c) and sustained-flux global warming potential (SGWP, Fig. 2d).

The annual CO_2 balance at our site range from the highest emissions of 2.44 ± 0.11 t CO_2-C ha^{-1} yr^{-1} in 2008 to the highest uptake of −1.25 ± 0.09 t CO_2-C ha^{-1} yr^{-1} in 2020, with a statistically significant decreasing trend of −0.36 t CO_2-C ha^{-1} yr^{-1} for the entire study period (Fig. 2a). In order to put the annual balances resulted from this study into perspective, we compared 52 site-years of EC data for different restored peatland sites and observed a wide range of annual balances from −4.46 ± 0.83 to 5.42 ± 0.41 t CO_2-C ha^{-1} yr^{-1}. While the uncertainty estimates in our gap-filled data range from 0.06 to 0.21 t CO_2-C ha^{-1} yr^{-1} for the annual CO_2 fluxes during the study period, the range of annual balance of CO_2 in our site still remains within the range of other published data from sites across all major peatland sites used in this review (Supplementary Refs 10–21 are used in the synthesis). When compared our annual values with the IPCC default Tier 1 EF of +0.50 (−0.71– +1.71) t CO_2-C ha^{-1} yr^{-1} and those suggested for the German NIR of −0.4 (−2.4 – +1.3) t CO_2-C ha^{-1} yr^{-1}, it becomes clear that our site has reached these EFs only 13 and 16 years after rewetting, respectively (Fig. 2a). This coincides with a change of the water level regime during the vegetation period (refer to the method section for vegetation period definition), when we observe the majority of the inter-annual variability in C balances. Starting from 2016 (the first dry year: based on a decreased rate of cumulative daily precipitation and series of consecutive days of
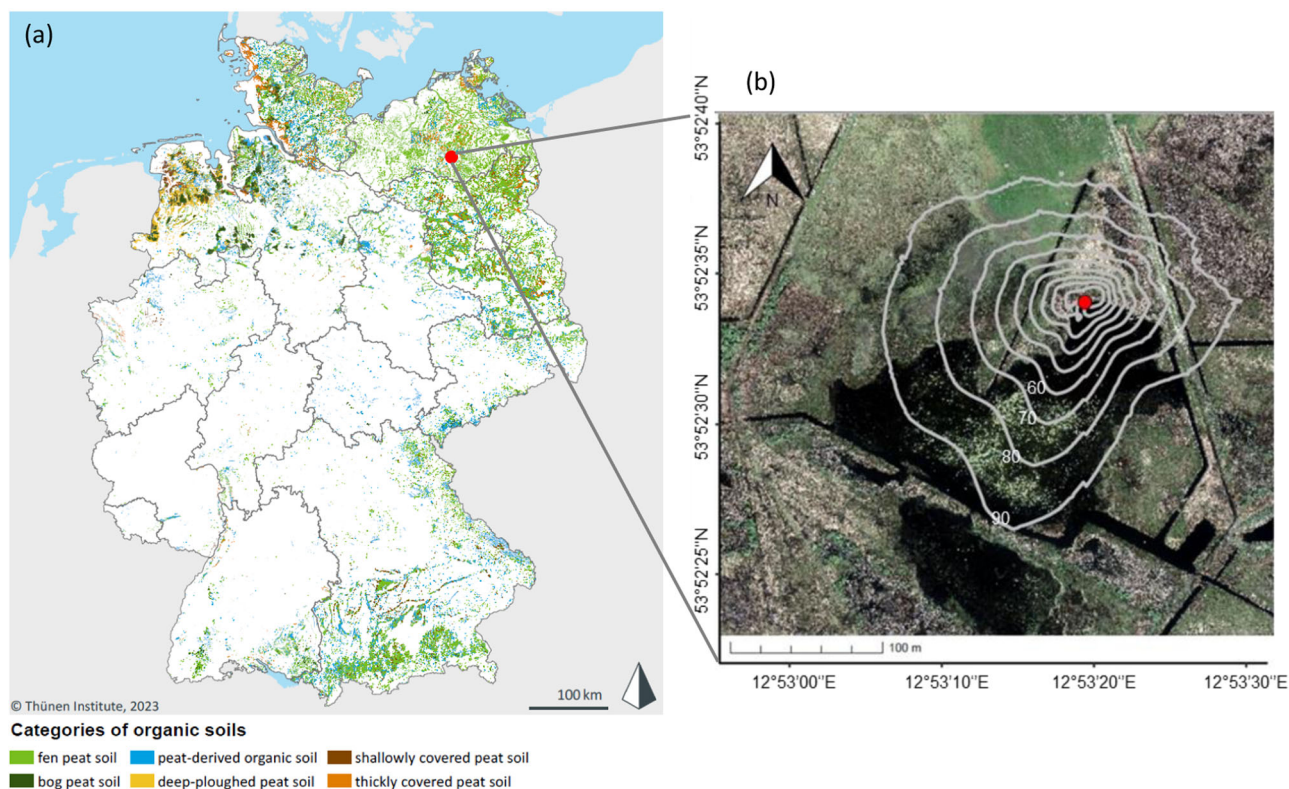
Fig. 1 Organic soils in Germany and the study site location. a Map of organic soils in Germany (updated map by Thünen Institute of Climate-Smart Agriculture, Braunschweig[66]). Estimate of organic soils is updated to 1.93 Mha (1.87 Mha without thickly covered peat soils) with legend units more relevant to GHG emissions, hydrological modeling and possible mitigation measures. **b** Location of the study site and footprint climatology calculated for the year 2018 using Kormann and Meixner[56] analytical model. Each isolines represent 10% of the annual flux footprint and the red dot indicates the location of the EC tower. The imagery used is a digital orthophoto image (resolution of 0.1 m) from 2018. Source: Landesamt für innere Verwaltung (LAiV), Mecklenburg-Vorpommern.

higher day-time temperature compared to the previous years), the monthly mean WTD during the vegetation period stayed mostly in the site-optimum range of being close to the surface or <0.3 m above the surface except for the year 2017 that experienced a wetter summer (Supplementary Fig. 2).

The annual CH_4 emissions at our site range from the highest emissions of 536.8 ± 8.45 kg CH_4 ha^{-1} yr^{-1} in 2008 to the lowest rate of 119.8 ± 0.72 kg CH_4 ha^{-1} yr^{-1} in 2019 with a decreasing trend of -23.6 kg CH_4 ha^{-1} yr^{-1} for the entire study period (Fig. 2b). Here, we also compared our CH_4 annual emissions with 25 site-years of EC data from published dataset for various restored peatland sites and observed a wide range of 65 ± 13 to 753.1 ± 24 kg CH_4 ha^{-1} yr^{-1}. While the uncertainty estimates in our gap-filled data range from 1.14 to 10.34 kg CH_4 yr^{-1} for the annual CH_4 fluxes during the study period, the range of our annual emissions of CH_4 still remains within the range of other published data across all other peatland sites used in this review (Supplementary Refs. 22–27 are used in the synthesis). Notably, in our case, the annual budget of CH_4 reached and fell below the IPCC default Tier 1 EF of 288 (0–1141) kg CH_4 ha^{-1} yr^{-1} and the ones suggested for German NIR of 279 (140–700) kg CH_4 ha^{-1} yr^{-1} only after the 2018 drought. A recent long-term study based on an interpolation of closed chamber measurements at the same site also reports a sharp decline in annual CH_4 emissions for 2016 and 2017, indicating the lowest annual post-rewetting emissions in 2017 (~12 years after rewetting)[32]. This agrees with our record up until that year showing the lowest annual emissions in 2017, compared to the previous years. Considering the drought effect and the possibility that CH_4 emissions may rebound to pre-drought levels, we determined trends over the observation period including and excluding the last three years and found the trend slope values of -23.6 and -16.8 kg CH_4 ha^{-1} yr^{-1}, respectively

(Fig. 2b). In order to estimate the future CH_4 emissions without the 2018 drought effect, we extrapolated pre-drought emissions based on the trend found for 2008–2018 period. We found that it would have taken approximately another 9 to 10 years (after 2017) for our site to reach the EF_{CH4} values of German NIR and IPCC Tier 1, respectively.

In order to compare the EF-projected and the observed GHG budgets, we calculated the cumulative GWP and SGWP as well as the cumulative emissions of CO_2 and CH_4 from our site for the consecutive 8-year measurement period of 2014–2021 via three different scenarios: 1- applying the IPCC default Tier 1 EFs; 2- applying the implied EFs suggested for the German NIR; and 3- annual budgets of CO_2 and CH_4 using our site-level EC-observed emissions (Table 1). Applying the default EFs given by the IPCC resulted in ~71% of the GWP and 76% of the SGWP estimated by the EC-site measurements. Applying the EFs suggested for the German NIR resulted in only 51% of the GWP and 57% of the SGWP calculated by the EC-site measurements. Evaluating the CO_2 and CH_4 emissions separately, we report here the 8-year cumulative CO_2 emissions via our EC measurements close to the IPCC Tier 1 scenario, albeit with high inter-annual variability. In terms of CH_4, while our annual EC emissions are within the range of EFs in the first two scenarios, the cumulative emissions based on IPCC and the German NIR are still at 72% and 70% of the cumulative emissions by our EC-site measurements, respectively (Table 1).

When comparing these scenarios, it should be noted that the individual data-points used in the derivation of the IPCC default EFs do not include sites with mean annual WTD of >20 cm above surface across the temperate climate zone (Fig. 3A.1b in ref. [5]). Similarly, the EFs for rewetted organic soils suggested for the German NIR were derived by applying the CO_2 response functions only for a WT range from -10 to 20 cm[8].
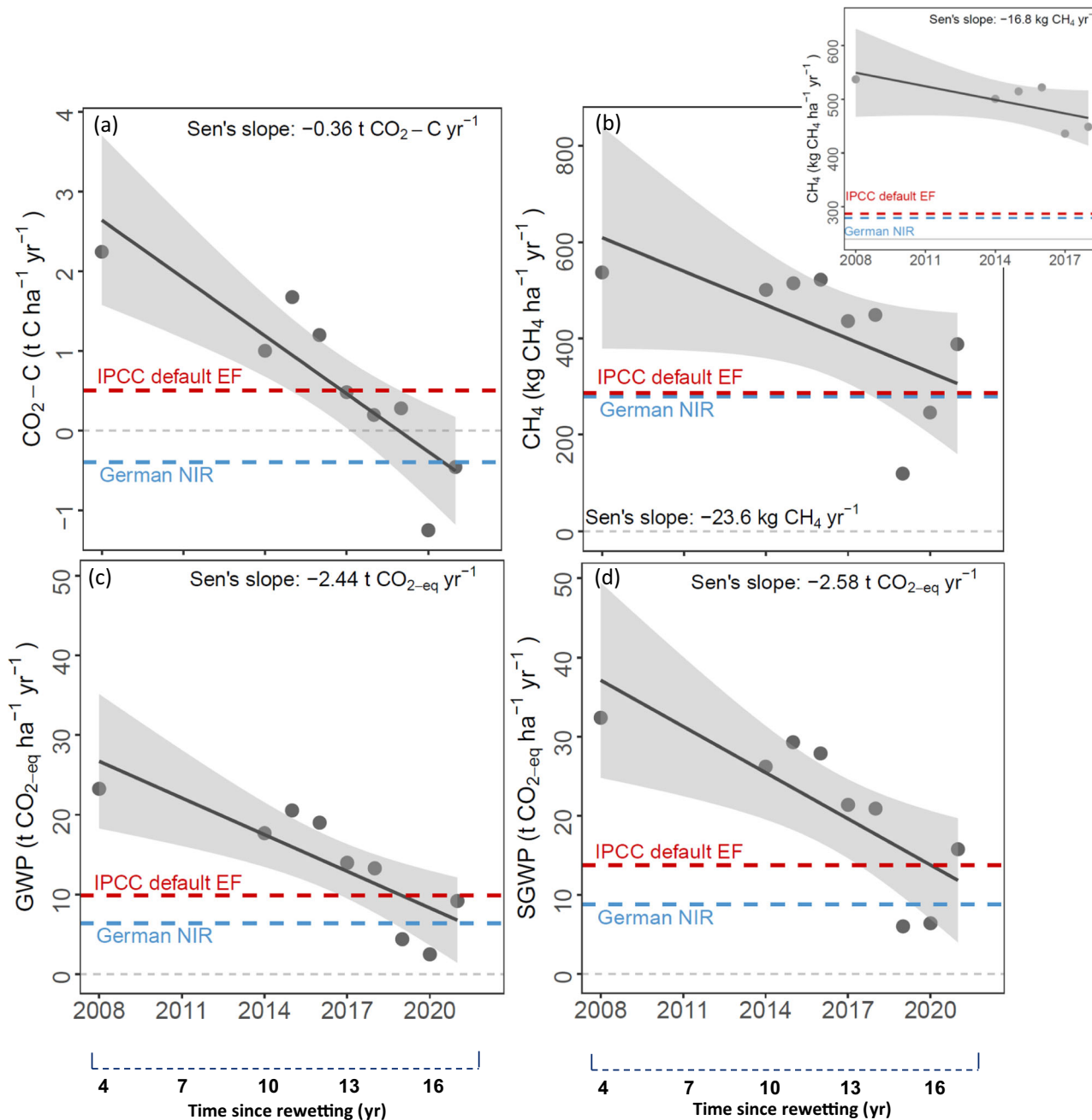
Fig. 2 Temporal trends and slopes of annual carbon balances throughout the study period. The observed annual balances of CO_2 and CH_4 and estimated GWP/SGWP at site-level measured by EC technique ($n = 9$ annual data, sum of 157824 half-hourly data for **a–d** and n = 6 annual data, sum of 105216 half-hourly data for the figure on the right corner of **b**). The IPCC default Tier 1 EFs and those suggested for the German NIR for rewetted organic soils compared with site-level measurements. The Theil-Sen estimate of slope is shown for each figure. The gray shaded area represents the 95% confidence interval around the regression line of best fit. The plots on the right corner of Fig. 2b refer to the Sen's slope of CH_4 trend prior to the period of impacted CH_4 fluxes due to the severe drought year 2018. CO_2: R^2: 0.78, p-value < 0.001. Sen's slope: −0.36 per year. CH_4: R^2: 0.41, p-value: 0.006. Sen's slope: −23.6 per year. GWP: R^2: 0.63, p-value < 0.001. Sen's slope: −2.44 per year. SGWP: R^2: 0.70, p-value < 0.001. Sen's slope: −2.58 per year.

Table 1 Cumulative CO_2 and CH_4 emissions for the consecutive period of 2014–2021.

Rewetted organic soils (temperate nutrient-rich)	CO_2 (t CO_2-C ha^{-1})	CH_4 (kg CH_4 ha^{-1})	GWP[a] (t CO_2-eq ha^{-1})	SGWP[b] (t CO_2-eq ha^{-1})
IPCC default (Tier 1) EF	4 (−5.68-13.68)[c]	2296.5 (0–6848)	79 (−20.7–241.8)	118 (−33.3–358.2)
German Implied EF	−3.2 (−19.2-10.4)[c]	2232.4 (1120–5600)	50.8 (−38.9–194.8)	88.7 (−70.2–290.05)
Site-specific (DE-Zrk), measured	3.12 (2.66–3.62)	3175.4 (2926.6–3438.3)	100.3 (91.6–109.5)	154.3 (144.9-164.4)

Comparison of the total 8-year period emissions; calculated emissions using the IPCC Tier 1 EFs, EFs suggested for the German NIR, and site-specific measured emissions (DE-Zrk).
[a]In order to calculate 100-year GWP estimates, values for annual CH_4 emissions are converted to the common CO_2-eq metrics of 27.2-28 (introduced by IPCC AR5, 5th assessment report (Myhre et al.[13]) and modified by the 6th assessment report).
[b]In order to calculate 100-year SGWP estimates, values for annual CH_4 emissions are converted to the CO_2-eq metrics of 45, when fluxes are sustained over time (estimated by Neubauer & Megonigal[57]).
[c]95% confidence interval is used to provide the 95% range.

The annual C balances presented in this study clearly show that rewetting started a process of transition that has already lasted for more than a decade. While high CH_4 emissions after rewetting are expectable, CO_2 emissions also continued on a level well above the currently used EFs. For both gases, a linear decline towards the emission factors suggested by the IPCC wetlands supplement can be approximated. We therefore conclude that the EFs provided by the IPCC and German NIR that essentially assign an abrupt step change of emissions to a peatland once its status changes from drained to rewetted, are not appropriate for capturing the C flux dynamics during the transition and thus overestimate the mitigation potential during at least the first decade after rewetting. Instead, we suggest to implement a temporal stratification that allocates a range of EFs according to the time since rewetting. An easy and very simple improvement would be using the existing EF values as fixed points at the time of rewetting and the (supposed) end of the transition period

and applying a linear decrease between those points. The length of the transition period should be chosen according to e.g., previous land use, WTD after rewetting, and vegetation development. The introduction of such a gradual transition from the EF for drained peatlands to the one for wet peatlands would already be valuable in reducing the error introduced to national inventory reports by the current approach. More long-term data from a variety of sites would help to derive robust estimates of the length of the transition period. Long-term data are also needed to better understand the drivers and controls of the emission dynamics during the transition period.

Drivers and controls on temporal variability of CO_2 and CH_4 exchange

Inter-annual variability in C fluxes. Inter-annual variability in C sink potential at our site is primarily driven by vegetation
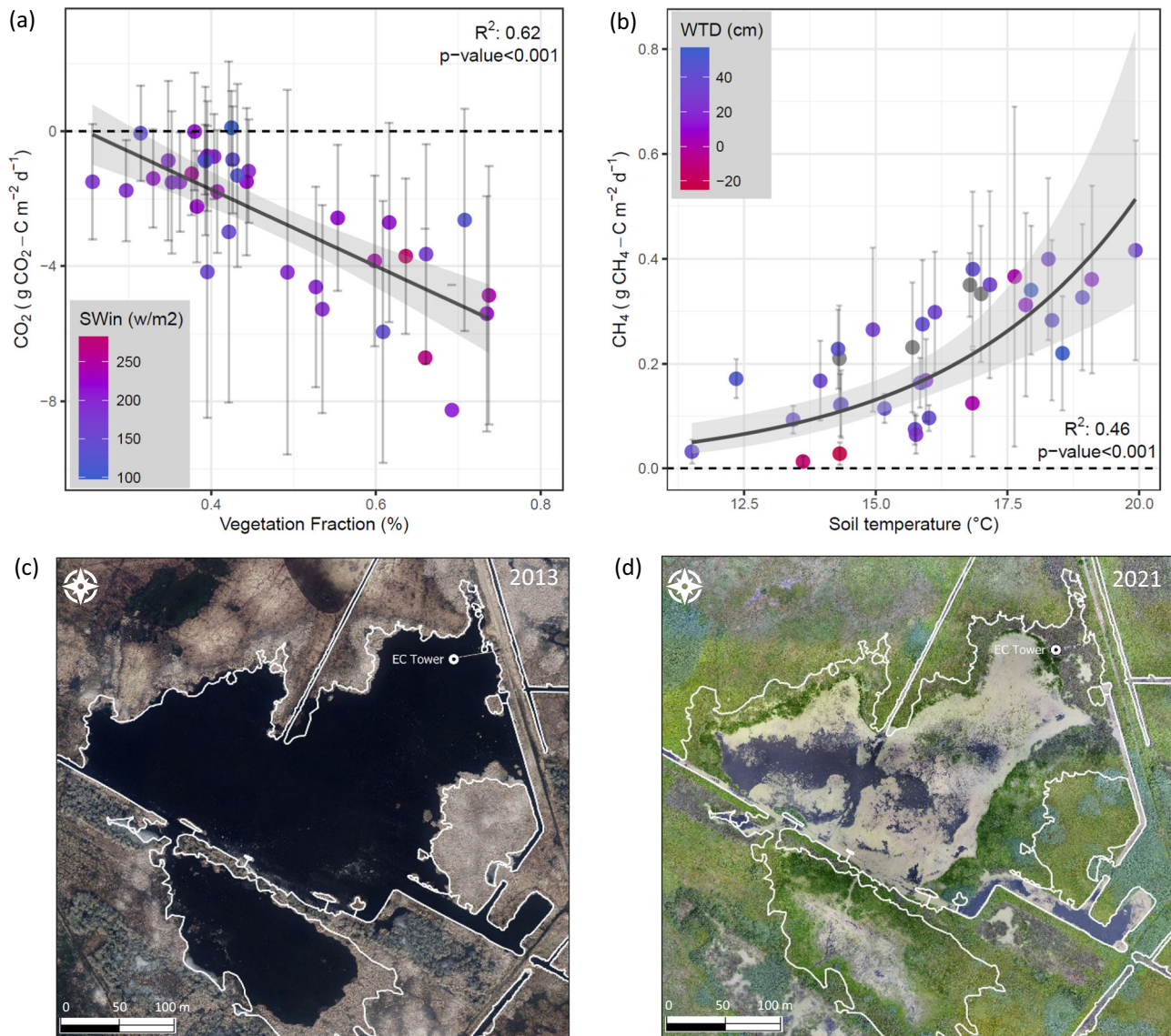

Fig. 3 Drivers and controls on temporal variability of CO_2 and CH_4 fluxes. The relationship between measured fluxdata (non-gap-filled data, where observation data were available) and other environmental variables during the vegetation period and vegetation development within the footprint of the tower during the study period: **a** monthly mean of NEE and vegetation fraction with fill colors according to the monthly shortwave incoming radiation ($n = 36$) and **b** monthly mean of CH_4 fluxes and soil temperature (at 10cm) with fill colors according to the monthly mean WTD ($n = 34$). Error bars denote standard deviation of the mean monthly values. **c** 2013 digital orthophoto image (source: LAiV) along with the outline of the open water area in 2010 overlaid on the DOP image. **d** 2021 UAV orthomosaic image (source: GFZ, section 1.4) along with the outline of the open water area in 2010 overlaid on the image showing the vegetation encroachment.

development and soil temperature. CO_2 fluxes during the vegetation period were predominantly controlled by vegetation fraction, i.e., the proportion of vegetation coverage within the footprint of the EC tower, explaining 62% of the variability (Fig. 3a). This follows post-rewetting successional vegetation dynamics which involved the establishment and growth of emergent helophytes (e.g., *Typha latifolia*) (Fig. 3c, d), increasing gross primary production (GPP) and ultimately causing the observed source-to-sink transition at our site (Fig. 2a). Even though WTD is generally considered the most important control on CH_4 fluxes, this is not the case at our site similar to the ref. [10] because of the predominant high water level (except for dry/drought periods). Instead, soil temperature is found to be the main driver of CH_4 emissions mainly during the pre-drought years (Fig. 3b). This finding agrees with the results by ref. [33] that identified soil temperature as the dominant control over wetland CH_4 fluxes at the seasonal scale. However, once WTD is below the surface (mainly during the drought and post-drought years), the temperature sensitivity is less pronounced, and the CH_4 emissions are affected by changes in the microbial community composition, resulting in decreasing emissions. When water level varies throughout the study period, WTD controls the temperature sensitivity of CH_4 fluxes. Ref. [34] also shows a lower water table is associated with a decrease in the temperature dependency of CH_4 emissions. Since the majority of CH_4 emissions primarily occur from April to September, the soil temperature during the vegetation period is still found to explain much of the variability (46%) of these emissions (Fig. 3b). Finally, a delayed recovery of CH_4 fluxes is observed after water table rise during the post-drought years. Peat core samples were analyzed in a study done by ref. [35] on microbial community dynamics at our site, and it is shown that the delayed recovery of CH_4 was due to the shifts in microbial community composition as a result of drought. They showed that the CH_4 cycling community is affected by drought by increasing the abundance of methanotrophs and diminishing the abundance of methanogens. Other studies[36,37] report similar findings regarding the higher concentrations of electron acceptors, thus changing redox conditions, during dry periods that may explain the delay in recovery of methanogenesis towards pre-drought levels.

Seasonal variability in C fluxes. Seasonal comparisons show that the long-term source-to-sink transition of CO_2 is directly related to greater GPP in periods of vegetative productivity (i.e., spring, summer, and early autumn period) while there was no major transition in non-vegetated periods (i.e., winter period) (Fig. 4a, c). Therefore, on average, >80% of the total annual CO_2 exchange corresponds to the vegetation period, with a relatively larger fraction in the last four years of measurements (Fig. 4). This may relate to a successional phase towards increased growth of emergent helophytes with relatively tall annual shoots and the drought-induced vegetation development within the footprint of the tower (Fig. 3c, d). Ten years after rewetting, gains in GPP began to offset losses due to ecosystem respiration (R_{eco}) during the vegetation period (Fig. 4b, c). Our time series data show that vegetation fraction within the tower footprint varied from 25–79% over the study period and that a strengthening of the net CO_2 sink during the vegetation period occurs with higher vegetation coverage (Fig. 3a). This was also highlighted in a recent study on a series of restored wetland sites in a Mediterranean climate[38]. If vegetation development continues, this would likely further bolster CO_2 sink strength. Furthermore, this may also soon represent the initiation of a post-rewetting steady-state if GPP growth continues surpassing growth in R_{eco}. Similarly, up to 84% of the total annual CH_4 emissions occurred in spring, summer, and early autumn period and minimal emissions were from the winter period, following a simple seasonal pattern

(Fig. 4d). Because methanogenesis is a strongly temperature-sensitive process, CH_4 emissions are relatively less substantial during the autumn and winter period, even though this is the period of annual peak water table (Supplementary Fig. 2). Emissions during the autumn periods following drier summers (2016, 2018 & 2019) were lower than those following summers with normal or higher WTD (Fig. 4d). This is linked to the lower WTD (Supplementary Fig. 2) and the lag effect from a dry summer leading to an increase in oxygenated water in the soil column and plausible continuous changes in microbial community dynamics during the autumn of the dry years.

Drought effects. Due to the importance of temperature and water level on total annual fluxes, drought-induced effects can lead to large inter-annual and seasonal variabilities. The results here show that our site transitioned into a consistent growing season CO_2 sink one year after experiencing dry conditions in summer 2016. In late summer of 2016, the water level was at the soil surface for the first time, and in 2018 a severe drought brought the water level down to well below the soil surface (WT < −30 cm) (Supplementary Fig. 2). Despite this, high plant productivity resulted in a net CO_2 sink during the summer seasons of the year 2017 and the following years because the deeper soil layers were still water-saturated and accessible to the vegetation. This is partially in accordance with the results shown in a recent study[16] indicating that high biomass production can compensate for decomposition losses in rewetted temperate fens during a dry year and may accumulate C during dry periods even when soils are not water-saturated. While our site was a net source of CO_2 during the vegetation period in the years prior to 2017 (Fig. 5a), this source already followed a decreasing trend. The dry conditions in 2016 and a drought in 2018 triggered the establishment of new vegetation types and spread of the existing vegetation to formerly open water areas during the vegetation period (Fig. 3d). This resulted in the increased rate of GPP outperforming R_{eco} and confirmed the persistence of the newly established vegetation after the drought event[31] and clearly led to an increased GPP in the following years in our site. This lag effect could highlight the resilience of CO_2 sink potential to drought conditions due to the drought-induced vegetation development for our site as well as other sites having capacity for further vegetation progress. In 2019, the CO_2 sink strength slightly decreased during the vegetation period which is strongly correlated with more cloudy days and less photosynthetic activity along with a higher respiration rate compared to the year before (Fig. 4a). In 2020, the whole ecosystem became a net annual CO_2 sink ($−1.25 \pm 0.09$ t CO_2-C ha^{-1} yr^{-1}) and this trend continued in the following year with a lower sink strength ($−0.46 \pm 0.07$ t CO_2-C ha^{-1} yr^{-1}). The drought in 2018 strongly affected CH_4 production potential in the anoxic soil layer within the drought period and during the following years (Fig. 5b), referred to as the lag effect. This is partially explained by the changes in the microbial community composition and described in detail in ref. [35]. While we found a strong reduction in CH_4 emissions in 2019 explained by the drought lag effect, the negative trend in CH_4 emissions in the years before the drought event is still significant ($−16.8$ kg CH_4 ha^{-1} yr^{-1}) (Fig. 2b). CH_4 emissions in 2020 and 2021 were higher than in 2019, but still continued decreasing from the pre-drought years and more closely following the pre-drought regression line (Fig. 2b).

Conclusions
Seventeen years after rewetting, our site is still in a transitional phase in which a CO_2 source-to-sink shift took place. While the CO_2 sink strength is increasing and annual CH_4 emission is
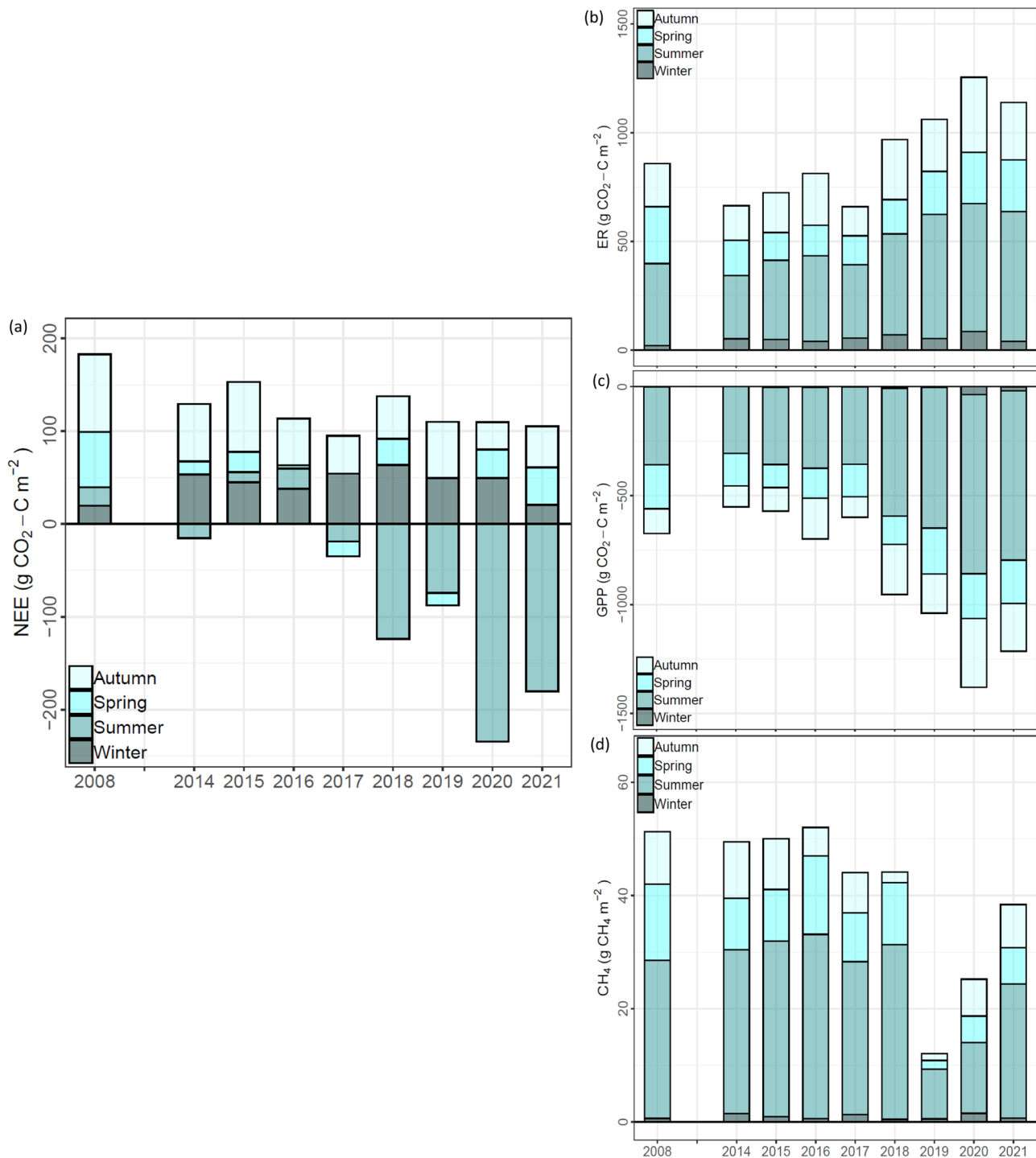
Fig. 4 Inter-annual and seasonal variability of CO$_2$ (NEE, R_{eco}, and GPP) and CH$_4$ fluxes. Seasonal variation of partitioned fluxes of CO$_2$; **a** net ecosystem exchange or NEE, **b** ecosystem respiration or R_{eco}, **c** gross primary production or GPP, and **d** seasonal variation of CH$_4$ emissions over the course of the study period.

declining, this confirms that the time needed for a rewetted ecosystem to return to its long-term CO$_2$ sink function may vary from years to several decades[39,40]. Though the ecosystem at our site has shifted to only a weak annual CO$_2$ sink during the last two years of measurements (2020: -1.25 ± 0.09 and 2021: -0.46 ± 0.07 t CO$_2$-C ha^{-1} yr^{-1}), the avoided CO$_2$ emission (over time) is already substantial when comparing to the net CO$_2$ emissions from drained temperate nutrient-rich sites (3.6 t CO$_2$-C ha^{-1} yr^{-1}: 1.8 – 5.4, 95% confidence interval, $n = 13$)[5]. Given the progressive vegetation development, we expect a further

increase in the net annual CO$_2$ sink strength until a steady-state is reached. Linear interpolation shows that if our site continues to follow its current CO$_2$ and CH$_4$ emission trajectories, a net cooling effect in terms of both GWP and SGWP can be expected within the next 4 and 6 years, respectively. Thus, the full potential of rewetting for climate mitigation may only be reached with a potentially long time-lag that depends on management practices before and after rewetting (e.g., site preparation and water level management) and environmental conditions. This time lag adds additional urgency to rewetting measures[20]. At the same time,
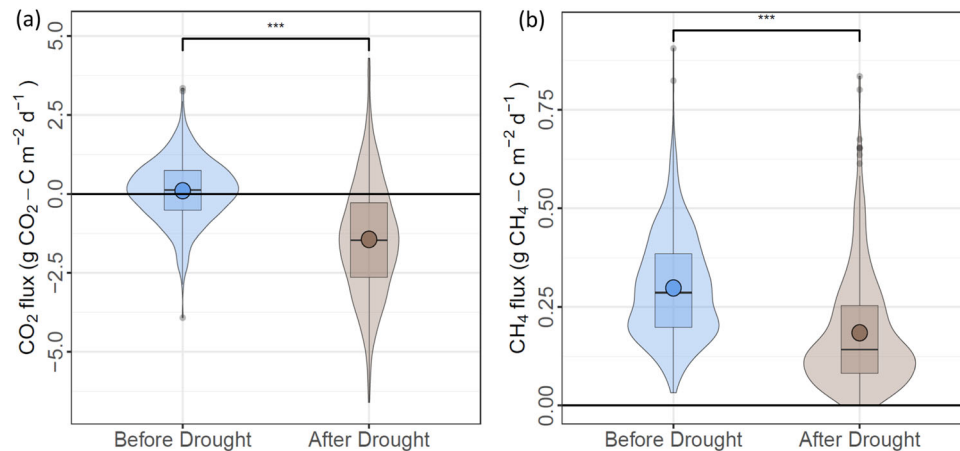
Fig. 5 Drought impact on CO$_2$ and CH$_4$ fluxes during the vegetation period. Blue Violin plots and their interquartile ranges indicate the probability distribution of CO$_2$ (**a**) and CH$_4$ (**b**) fluxes during the vegetation period before 2018 drought ($n = 720$ for 2008 and 2014–2017 period). The brown Vplots indicate the probability distribution of fluxes during and after 2018 drought ($n = 576$ for 2018–2021 period). The dots in both plots represent the median of the daily cumulative GHG fluxes. Asterisks represent the significance value, ***$p < 0.001$.

given the potentially long transition period, much longer post-rewetting monitoring is needed at a representative selection of sites in order to cover the GHG source-to-sink transition, capture the impacts of extreme events[41], and better project emission trajectories in a warmer and potentially drier future.

Methods

Site description. Here, we present a long-term year-round CO$_2$ and CH$_4$ flux dataset from the Zarnekow peatland site (53°52.34' N, 12°53.21' E; Fluxnet-ID: DE-Zrk) in northeast Germany (Fig. 1), representing, to our knowledge, the longest time since rewetting (17 years). The EC measurements at the site initially ran from autumn 2007 through spring 2009 followed by a four-year break before being resumed in spring 2013. Since then, measurements have been continuous. This study includes data until the end of year 2021. The Polder Zarnekow is a rich minerotrophic percolation fen within the Peene river mire system. Drainage activities in this region started in the 18th century and intensified between 1960 and 1990 for grassland use[42,43]. Starting in late 2004—early 2005, our site was rewetted by discontinuing drainage activities, including dismantling of an active pumping station[31,44] while retaining the dikes until they fail naturally. This resulted in water levels permanently above the ground surface. Following the rewetting activities, our site turned into a spatially heterogeneous site consisting of emergent vegetation and open water areas. The former grassland was initially dominated by reed canary grass (*Phalaris arundinacea*). During the first year of inundation, the reed died off, and a new sediment layer that had a high content of relatively fresh plant litter was formed[45]. Since the second year of inundation, the water body of the shallow lake has been dominated by aquatic plants like *Ceratophyllum demersum* and *Lemnaceae* before emergent macrophytes invaded[45]. Afterwards, Helophytes (e.g., *Glyceria maxima* and *Typha*) started the colonization of lake margins. The perennial herbaceous plant *Typha*, or common cattail showed an increasing proportional coverage in the latest years while *Sparganium erectum* was detected as the first vegetation type to colonize the open water in the years after rewetting. Starting from 2016, *Sparganium* seemed to invade farther into the open water than before, however, in 2018 and 2019 they did not cover the entire shore, but grow only in some patches[46]. *Phragmites australis*, as a potentially peat-forming species, has no substantial expansion or invasion; thus, it plays a minor role in the vegetation succession in the Polder Zarnekow so far[46]. The mean annual air temperature at our site is 9.7 °C and the WTD varied from ~ −30–120 cm throughout the study period. The average annual precipitation is 544 mm, while the peat depth is estimated at ~10 meters[42,47].

Representativeness of the study site. Fen peat soils, such as at our study site, represent the category of organic soils with the largest areal coverage in Germany, particularly in northeastern Germany (47.4% versus bog peat soils (13.9%), peat-derived organic soils (24.4%), shallow (10 to <30 cm; 4.6%) and thickly (30 to <100 cm; 3.4%) covered peat soils as well as deep-ploughed peat soils (6.4%), Fig. 1a). While the draining system, ditches, and pumping stations were established in a similar way[24] for the majority of the peat soils, the rewetting methods are quite diverse nationwide[24]. In eastern Germany, many of the already rewetted peatlands were taken out of agricultural use and water levels were not actively managed, especially in Mecklenburg-Vorpommern and Brandenburg[24]. While the above-ground water levels vary, our study site is a good example of this category with previous grassland use, abandonment, and eventual flooding. We therefore consider our study site representative of peatlands with higher mean annual WTD (30–50 cm), especially in the river valley mire systems in northeastern Germany.

Eddy covariance measurements and uncertainty estimates. Year-round CO$_2$ and CH$_4$ flux measurements were conducted using the EC technique at the ecosystem scale. Fluxes from both open-path and enclosed-path gas analyzers were processed with an averaging interval of 30-minutes using the EddyPro software (LI-COR Inc. Lincoln, Nebraska, USA). CH$_4$ exchanges and NEE, involving both CO$_2$ uptake processes by plants during photosynthesis (GPP) and CO$_2$ release processes through plant and soil respiration (R_{eco}), are continuously monitored. The detail of instrumentations, flux processing, and quality control steps are described in the Supplementary Methods. The uncertainty estimates for gap-filled annual and seasonal sums were calculated from the 95% confidence intervals of the annual sums of 20 iterations of random forest (RF) imputed data for both CO$_2$ and CH$_4$ fluxes. The random measurement errors were also calculated for both gases and for both enclosed and open-path analyzers, but they were not included in the total estimated uncertainties in the case of CH$_4$ fluxes due to the values being negligible.

Gap-filling methods to estimate annual budgets of CO_2 and CH_4 fluxes and partitioned NEE. Missing and bad quality data–due to instrument failure and implausible spikes, power outage, and precipitation (for open-path gas analyzers)—are common in biogeochemical studies. In this study site, the majority of missing or removed values were from nighttime data and mainly due to low turbulence. Also, power shut-downs and instrument failure led to a one-time maximum data loss of seven continuous weeks in the early period of the study. Gap-filling methods based on machine learning algorithms, particularly using RF, are gaining acceptance as a benchmark for non-parametric imputation methods. There are few studies on widely established gap-filling methods[48–52], specifically for CH_4 fluxes[53]. It is also suggested that deploying multiple gap-filling techniques will lead to a more robust flux aggregation and estimation of the uncertainties[52]. Hence, we completed a cross-comparison study here using three different gap-filling methods by creating artificial gap scenarios (short and long gaps) within different seasons and training various gap-filling models. The three tested methods are marginal distribution sampling (MDS) as a multi-step look-up table, and two machine learning algorithms i.e., artificial neural network (ANN), and RF. We further cross-validated our results and found that the decision tree algorithms based on RF outperforms other methods. Finally, the RF imputed CO_2 fluxes were partitioned into R_{eco} and GPP by calculating R_{eco} for all half-hourly periods using REddyProc package[54,55]. Seasons are defined as meteorological seasons based on the annual temperature cycle and the Julian calendar days in the northern hemisphere. Meteorological spring includes March, April, and May (M-A-M); summer includes June, July, and August (J-J-A); autumn includes September, October, and November (S-O-N); and winter includes December, January, and February (D-J-F). In this study, the vegetation period is based on a 10 °C threshold of monthly average soil temperatures at 10cm depth and lasts from May to September in most of the years. More details of gap-filling and partitioning methods are described in the Supplementary Methods.

Accounting for spatial heterogeneity in flux gap-filling and estimating the vegetation fraction within the tower footprint. In order to correct the heterogeneity-impacted flux measurements, the half-hourly flux footprints were calculated using the analytical footprint model of Kormann and Meixner[56] (Fig. 1b). We further calculated the flux contribution for each half-hourly footprint for a 1-meter grid extending 1000 m north, south, east and west of the EC tower for different land surface cover using high-resolution imagery. The classification was done by a manual delineation of the open water areas using the software QGIS (version 3.16.11-Hannover). The aim was to determine the spatial coverage of open water versus wetland vegetation during the peak vegetation period for each year. The resulting classified images are centered on the EC tower, with a ground size of $2 \times 2\,km^2$ and a pixel size of $1\,m^2$. By overlaying the annual land cover classifications on our footprint models, the percentage of different classes in the footprint of the tower was estimated and finally used in the RF algorithms to improve our flux gap-filling. Determination of the fraction of signal originating from different land surface classes improved the imputation procedure in estimating annual emissions with lower root mean square error (RMSE) and higher R^2 (Supplementary Fig. 1). The land surface cover is categorized into three different classes in this study: 1- open water, 2- dense emergent wetland vegetation which is dominated by perennial herbaceous plant such as cattail and reeds, and 3- (managed-dry) grassland vegetation (in the sector NW-NE of the EC tower). The details of methods involved in the contribution of land cover classes to half-hourly fluxes are described in the Supplementary Methods.

CO_2 and CH_4 emission factors in rewetted peatlands. The resultant time series and calculation of the relative contribution of each surface type to corresponding measured fluxes are considered to be essential in reducing uncertainty with regard to spatio-temporal heterogeneity in our site. This led to an improved estimation of annual C balances, and accordingly GWP and SGWP estimates for a rewetted peatland fen site representing the nutrient-rich organic soils in temperate climate zones. In order to assess the minimum and maximum climatic effect, we took the slope of annual CO_2 emissions for the entire study period and the slope of CH_4 emissions with and without the 2018 drought effect (-23.6 vs. $-16.8\,kg\,CH_4\,ha^{-1}\,yr^{-1}$). Following the trajectory prior to the drought year, the maximum timeline to reach a net cooling effect is estimated for 2027 and 2029 for GWP and SGWP, respectively. In order to determine the magnitude of the climatic effect of the ecosystem, we report both GWP (considering a single pulse emission for CH_4) and SGWP (considering CH_4 fluxes persist over time[57]).

The EFs introduced by the IPCC 2013 Wetlands Supplement[5], as the most robust meta-study available, and the ones suggested for the German NIR[8] provide initial basic estimates of EF_{CO2} and EF_{CH4} globally and nationally, respectively. The global default Tier 1 EFs delivered by the IPCC 2013, for temperate rewetted nutrient-rich sites, are estimated based on closed chamber and EC measurements. Similarly, the EFs suggested for the German NIR for rewetted organic soils were estimated based on a dataset from 39 chamber measurements sites across 11 study areas (maximum duration of measurements: 5 years) with water tables below $-0.1\,m$ or close to the surface (Table S2 in ref. [8], SI2). Thus, the uncertainty associated with the individual data-points used in the derivation of the default Tier 1 EFs is considerable as the compiled studies are generally based on short-term (1–2 years) datasets[5]. Temporal trends of C-exchange during the transitional phase that rewetted sites experience until a new steady-state is established are not fully detected within these short datasets (IPCC Methodologies[5]: Annex 3 A.1. for CO_2 and 3 A.3. for CH_4). Also, data from undrained organic soils were used as proxy for rewetted organic soils for the derivation of EFs in the IPCC 2013 Wetlands Supplement[5]. The same approach was applied in a previous study[10] on the derivation of EFs for rewetted organic soils and in ref. [8] for suggested EFs in the German NIR. The suggested EF_{CO2} for the German NIR is estimated by applying the CO_2 response function for the WTD range from $-0.1\,m$ to $0.2\,m$ (in analogy with IPCC 2013 Wetlands Supplement). The EF_{CH4} (only land) suggested for the German NIR was also derived by applying the response functions to water table. Therefore, we anticipate that using more long-term datasets of both in-situ ecosystem-scale measurements and high-resolution remote sensing monitoring products would partially overcome the uncertainties of the implied EFs based on response functions to WTD.

Statistical and data analysis. The estimates of the slope and the trend for annual emissions of CO_2 and CH_4 over the course of the study period are computed using the Theil-Sen approach via 'zyp' R package[58]. A Mann–Kendall test is used to assess the significance of the trend in the time series analysis and to obtain p-values. RMSE was calculated for comparisons of imputed data with and without the inclusion of vegetation fraction in the training data using the 'Metrics' R package[59]. Univariate linear regressions were used to examine the relationship of fluxes with environmental data (Fig. 3). The regression of CH_4 fluxes with water table was performed with log linearized data due to the skewed distribution of the fluxes (Fig. 3a). We used the 'missForest' R package[60] for the imputation of half-hourly CO_2 and

CH_4 fluxes and the 'REddyProc' package[55] for partitioning NEE into GPP and R_{eco}. Statistical analyses were performed in R software[61]. Data organization was performed using the 'data.-table' R package[62]. Land cover classification was performed in QGIS, v.3.16.11-Hannover. Footprint calculation was done in MATLAB 2020a, the MathWorks Inc. Natick, Massachusetts, United States. Further analyses and visualization of the data were done using the 'ggplot2' R package[63].

Data availability

Code availability

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Author contributions

A.K. led the study design and the writing of the article with contributions from all other authors. Contributions from authors are listed below: A.K.–conceptualization, methodology, software, formal analyses, validation, visualization, writing–original draft, review, and editing. C.W.–data acquisition and eddy covariance raw data processing, methodology, software, writing, review, and editing. P.G. and Z.L.–writing, review and editing. J.H.–software, writing, review, and editing. K.K.–software (land surface classification). T.S.–conceptualization, funding acquisition, data acquisition, investigation, writing, review, and editing. All authors contributed to data interpretation.

Competing interests

The authors declare no competing interests.

Additional information

Supplementary information The online version contains supplementary material available at https://doi.org/10.1038/s43247-024-01226-9.

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Peer review information *Communications Earth & Environment* thanks Pierre Taillardat and the other, anonymous, reviewer(s) for their contribution to the peer review of this work. Primary Handling Editors: Mengze Li, Clare Davis, Aliénor Lavergne. A peer review file is available.

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