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RESEARCH ARTICLE

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Key Points:

- $\delta^{13}\text{C}$ - CH_4 and $\delta^2\text{H}$ - CH_4 of peatland emissions are both variable and show opposing relationships with CH_4 fluxes
- Combined $\delta^{13}\text{C}$ and $\delta^2\text{H}$ patterns indicate a seasonal shift from hydrogenotrophic to acetoclastic methanogenesis as CH_4 emissions increase
- CH_4 emissions were highly variable and positively correlated with temperature, dissolved organic carbon, and soil water color

Supporting Information:

Supporting Information may be found in the online version of this article.

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Seasonal and Interannual Variations in Methane Emissions and Carbon and Hydrogen Isotopes From an Upland Ombrotrophic Peat Bog

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Abstract Despite their importance for global CH_4 biogeochemistry, there is limited evidence on seasonal patterns in CH_4 stable isotope composition from peatlands. We conducted a 2.5-year study investigating seasonal changes in CH_4 fluxes and dual-stable isotopes ($\delta^{13}\text{C}$, $\delta^2\text{H}$) in an ombrotrophic peat bog in Northern England. We performed chamber flux measurements and incubation experiments to quantify CH_4 and CO_2 fluxes, CH_4 isotopes, and CH_4 production and oxidation potential. CH_4 fluxes varied significantly across seasons and years, with exceptionally high emissions ($>45,000 \mu\text{mol m}^{-2} \text{h}^{-1}$) recorded in summer 2016, contrasting with median fluxes of $10.6\text{--}91.8 \mu\text{mol m}^{-2} \text{h}^{-1}$. Fluxes correlated with soil temperature and soil solution DOC and color, but not with water table depth, which had limited variability. Laboratory CH_4 production potentials peaked below the water table at 30–40 cm depths, while oxidation potential decreased with depth. Carbon and hydrogen isotope δ -values ranged from -88.4‰ to -73.2‰ and -345.3‰ to -275.6‰ , respectively and showed opposing trends as $\delta^{13}\text{C}$ increased with CH_4 flux while $\delta^2\text{H}$ decreased. This is consistent with increasing fluxes being driven by acetoclastic methanogenesis. Our results highlight that seasonal shifts in CH_4 emissions are accompanied by changes in isotopic signatures, even without major changes in water table or CH_4 oxidation. Incorporating temporal variability into isotope-based CH_4 source attribution may therefore improve constraints on peatland emissions in global CH_4 budgets.

Plain Language Summary Peatlands are large sources of the potent greenhouse gas methane (CH_4). Understanding the conditions that affect the release of CH_4 from peatlands is key to understanding the global CH_4 cycle. Measuring ratios of C and H isotopes in atmospheric CH_4 is commonly used to attribute emissions as they depend on the pathway by which the methane was generated. The isotope ratios of peatland CH_4 emissions are often assumed to be constant over time. We measured CH_4 fluxes and their stable carbon ($\delta^{13}\text{C}$) and hydrogen ($\delta^2\text{H}$) isotopes over 2.5 years from a peat bog. CH_4 emissions varied strongly across seasons, including exceptionally high fluxes during summer 2016. CH_4 emissions correlated with soil temperature and soil water chemistry but not water table depth, which remained relatively stable. The carbon and hydrogen isotope values showed opposite trends: $\delta^{13}\text{C}$ increased and $\delta^2\text{H}$ decreased as CH_4 fluxes increased. This indicates that higher emissions were mainly driven by the conversion of acetate to CH_4 . Our findings show seasonal changes alter both CH_4 emissions and their isotopic fingerprints, even when water tables and therefore oxygen availability are relatively stable. This suggests that considering seasonal variability in isotopes could improve the accuracy of global peatland CH_4 emission estimates.

1. Introduction

Natural wetlands are the largest source of methane (CH_4) emissions globally, estimated to account for around 20%–30% of total emissions (Kirschke et al., 2013; Saunio et al., 2025), and their contribution is projected to increase considerably under future climate scenarios (Dean et al., 2018; Koffi et al., 2020; Zhang et al., 2017). Despite a solid body of research, their contribution to changes in the global CH_4 budget over the past decades remains ambiguous, with large uncertainties in emissions (Saunio et al., 2025; Turner et al., 2017), though recent studies indicate that their contribution to total emissions has been increasing (Ciais et al., 2026; Drinkwater et al., 2023; Oh et al., 2022; Peng et al., 2022). Part of this uncertainty is due to the large seasonal and inter-annual variability in emissions as well as high spatial heterogeneity (Lai, 2009; Limpens et al., 2008; Turetsky et al., 2014; Yuan et al., 2024). Stable isotope analysis is commonly employed to constrain regional to global scale

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methane budgets (e.g., Deng et al., 2017; Dlugokencky et al., 2011; Kirschke et al., 2013; Lan et al., 2021; Oh et al., 2022; Umezawa et al., 2012) as source categories differ in their $^{13}\text{C}/^{12}\text{C}$ and $^2\text{H}/^1\text{H}$ ratios (Levin et al., 1993; Sherwood et al., 2017; Whiticar, 1999). However, seasonal variation of stable isotope emission signatures is only rarely incorporated into current biogeochemical models (e.g., Oh et al., 2022).

Peatland CH_4 fluxes and their isotope signatures are determined by a complex interplay of soil, meteorological, vegetation, and landscape factors. Compared to CO_2 fluxes, they are considered poorly predictable on local to global scales. Water table depth (WTD) and soil moisture are known to strongly affect CH_4 emissions as they determine the depth of the oxic and anoxic zones of the peat (Basiliko et al., 2007; Smith et al., 2003), and therefore, microbial CH_4 production and oxidation at different depths. Lower WTD may also lead to an enrichment in CH_4 isotope signatures due to isotopic fractionation during CH_4 oxidation in more aerated peat. Temperature is another key driver of CH_4 , with higher temperatures promoting microbial methanogenesis across a wide range of ecosystems (Yvon-Durocher et al., 2014). Temperature is also thought to change the relative activity between microbial CH_4 production and oxidation, with methanogens being more temperature sensitive than methanotrophs (Le Mer & Roger, 2001; Winden et al., 2012), thus potentially not only altering net emissions but also shifting CH_4 isotope signatures. Vegetation cover also influences CH_4 fluxes, as it affects both inputs of carbon (C) into the peat and the potential pathways of CH_4 release into the atmosphere. In addition to diffusive release into the atmosphere and ebullition through bubbles, CH_4 can pass through the aerenchyma of many wetland vascular plants (e.g., sedges), which act as conduits for gases (Bridgman et al., 2013). Plants also provide labile C to soil microbes, thus increasing microbial activity, changing microbial substrates, and microbial communities (Chowdhury & Dick, 2013).

Microbial CH_4 in wetlands is produced by methanogenic archaea by one of two main metabolic pathways: acetoclastic methanogenesis, using acetate as substrate, and hydrogenotrophic methanogenesis, using CO_2 and H_2 as substrate. Both forms occur in freshwater wetland systems, with acetoclastic methanogenesis considered to dominate CH_4 emissions (Kotsyurbenko et al., 2004; Whiticar, 1999). These two pathways have distinct stable isotope signatures, with CH_4 formed through acetoclastic methanogenesis being more enriched in ^{13}C and more depleted in ^2H ($\delta^{13}\text{C} = -60\text{‰}$ to -50‰ and $\delta^2\text{H} = -400\text{‰}$ to -250‰) compared to hydrogenotrophic methanogenesis ($\delta^{13}\text{C} = -110\text{‰}$ to -60‰ and $\delta^2\text{H} = -250\text{‰}$ to -170‰) (McCalley et al., 2014).

Northern peatlands are a major contributor to global wetland CH_4 emissions (Melton et al., 2013). Their stable carbon isotope values are typically in the range of -59‰ to -74‰ $\delta^{13}\text{C}$ (Feinberg et al., 2018). However, few studies have investigated their seasonal variations and most studies are limited to a few samplings. Chasar (2000) found variations in isotopic values from June–September in a Minnesota peat bog but not a fen. In Finnish wetlands, Fisher et al. (2017) have found increasingly lower values from summer to autumn, while Sriskantharajah et al. (2012) have found slightly higher values after the spring melt. Fluctuations throughout the course of a growing season were observed by McCalley et al. (2014) using continuous measurements in a Swedish bog and fen. The few studies that have measured $\delta^2\text{H}$ in peatlands included only limited samplings and did not investigate seasonal trends. Reported values of direct measurements include -300‰ for a peat bog in West Virginia (Wahlen et al., 1990), and $-338.5 \pm 8.3\text{‰}$ for a peat bog in North West England (Takriti et al., 2023), while Menoud et al. (2022) reported values of $-346 \pm 6.7\text{‰}$ for polar swamps and $-271 \pm 4.4\text{‰}$ for a temperate swamp in Sweden. Atmospheric models usually do not include $\delta^2\text{H}$ data, even though they may help to constrain source estimates (Dlugokencky et al., 2011; Ganesan et al., 2019; Mannisenaho et al., 2023). Moreover, isotopic measurements in wetlands are often limited to the growing season, while nongrowing season CH_4 emissions can contribute significantly to total annual emissions (Treat et al., 2018). As stable isotopes are key for tracing the sources of CH_4 emissions to the atmosphere, this lack of understanding limits the ability to constrain peatland emissions and understand their responses to environmental drivers.

The overarching aim of this study was to investigate temporal changes in stable isotope ratios of peatland CH_4 fluxes. Specifically, we combined CH_4 flux measurements with dual stable isotope analysis as well as laboratory incubations of peat soils to (a) investigate seasonal variations in wetland $\delta^{13}\text{C}\text{-CH}_4$ and $\delta^2\text{H}\text{-CH}_4$ emission values; (b) investigate relative changes in microbial CH_4 production and oxidation inferred from isotope data and laboratory incubations in response to environmental drivers. We hypothesized that (a) there would be a seasonal pattern in flux rates driven by changes in temperature and water levels; (b) CH_4 production would be more temperature sensitive than CH_4 oxidation and $\delta^{13}\text{C}\text{-CH}_4$ and $\delta^2\text{H}\text{-CH}_4$ would therefore be positively correlated with peat temperature; (c) Decreasing water table levels would increase CH_4 oxidation and lead to an enrichment

in both $\delta^{13}\text{C}\text{-CH}_4$ and $\delta^2\text{H}\text{-CH}_4$ values. We also investigated the relationship between CH_4 and CO_2 , that is, ecosystem respiration (R_{eco}) as a more readily available indicator of carbon cycling and decomposition processes in peatlands.

2. Materials and Methods

2.1. Study Area

The study was undertaken at a temperate ombrotrophic blanket peat bog in Northern England, UK ($54^\circ 41' 44.2''\text{N}$, $2^\circ 23' 16.5''\text{W}$) within the Moor House National Nature reserve. Long-term-monitoring environmental data are available for the site through the UK Environmental Change Network (ECN) (Rennie et al., 2020), and it has been shown to be an important source of CH_4 emissions in the UK (McNamara et al., 2008; Ward et al., 2007, 2013).

The mean annual temperature and precipitation are 5.8°C , and 2,148 mm, respectively (UK Environmental Change Network). The nature reserve has an altitude ranging from 290 to 850 m with the study site located on a relatively water saturated level area at an elevation of 565 m. It is acidic (pH of 3.0–4.2), oligotrophic, and ombrogenous, receiving water and nutrients from rainfall. Mean peat depth in the area is around 2 m (Rose et al., 2016). The vegetation community is classified as a *Calluna vulgaris*–*Eriophorum vaginatum* blanket mire, *Empetrum nigrum* ssp. *nigrum* sub-community M19b according to the UK National Vegetation Classification (NVC) (Rodwell, 1993). Plant species composition at each flux measurement location was determined as the approximate area of coverage according to the DAFOR scale (Hurford et al., 2006), see Table S1 and Figure S2 in Supporting Information S1.

2.2. Flux Measurements

In June 2015, five locations were randomly selected from an area measuring around 100 m^2 for flux measurements (Figure S1 in Supporting Information S1). A gas sampling collar, 30 cm in diameter and 10 cm high, was permanently installed in each location (1–5). Collars were placed onto the soil and inserted up to a depth of around 5 cm into the uneven surface to form a level base. Flux measurements of CH_4 and CO_2 , that is, ecosystem respiration, were taken weekly from June 2015 to October 2016, after which measurements continued every 2 weeks until October 2017. Flux chamber measurements were conducted using a Los Gatos Ultraportable Greenhouse Gas Analyzer (UGGA; Los Gatos Inc., San Jose, USA): Collars were sealed with opaque 19 L dome-shaped chambers and the CH_4 and CO_2 concentrations in the chamber were measured for a minimum of 5 minutes. Flux rates were calculated from the rate of gas concentration change as described by McEwing et al. (2015) based on a minimum of 30 s of continuous measurements. To ensure consistent treatment and quality of data, we adapted the DEFRA SP1210 protocol for the analysis of flux chamber data (Evans et al., 2017): Linear regressions of CH_4 or CO_2 over time were excluded from further analysis if they had both an $R^2 < 0.7$ and a change in concentration of $>0.01\text{ ppm}$ and $>1.5\text{ ppm}$ for CH_4 and CO_2 , respectively. Measurements with a lower change in concentration were not excluded as R^2 approaches 0 as the flux approaches 0. In addition, measurements visibly influenced by ebullition events or technical errors were excluded. After quality control, 371 measurements of CH_4 and 349 measurements of CO_2 fluxes were included in our analysis.

2.3. Isotope Measurements

Isotope samples were collected in September 2015, monthly from January 2016 to January 2017, and additionally in March and October 2017 on a total of 16 dates. At the end of the flux measurements, chambers were left on the collars for a total of 1 hr to allow gas concentrations to build up to the necessary concentration for isotopic analysis. After 1 hr, 20 mL samples of headspace gas were extracted from each chamber with a syringe through septa and filled into evacuated 12 mL Exetainer vials (Labco Ltd., Ceredigion, UK) for C and H isotope analysis. A further 10 mL of gas was added to 3 mL Exetainer vials for CO_2 and CH_4 concentration analysis. Syringes were purged with ambient air between each sample. Stable isotope analysis of CH_4 was performed at the UC Davis stable isotope facility as described in Yarnes (2013). CH_4 concentrations were measured by gas chromatography using an Autosystem XL GC (Perkin Elmer, Waltham, MA, USA) as described in Case et al. (2014). CH_4 concentrations were too low for H isotope analysis at six time points, which were excluded from further analysis. All stable isotope values reported in this study are presented using the δ -notation, in per mille (‰), relative to the international standards Vienna Pee Dee Belemnite (VPDB) for C and Vienna Standard Mean Ocean Water (VSMOW) for H (Takriti et al., 2023).

2.4. Soil Moisture, Water Table Depth, Temperature, and Water Chemistry

Soil physical parameters, soil solution chemistry, and meteorological data were measured as part of a long-term monitoring program by the UK ECN in a topographically similar area around 20 m away from the flux chambers (Figure S1 in Supporting Information S1). For this study, we used ECN data on air and soil temperature, and for exploratory analysis, soil solution pH, dissolved organic carbon (DOC), and soil solution color. For a detailed description of the data, see Rennie et al. (2020). In short, soil temperature was recorded using 108-LC temperature probes (Campbell Scientific, Loughborough, UK) at 10 and 30 cm depth. Samples for soil solution DOC and soil solution color from 10 to 50 cm depth were collected and analyzed every 2 weeks. DOC concentration was measured by combustion using a TOC-L (Shimadzu, Tokyo, Japan) and water color by absorbance using a Cary 60 UV-Vis Spectrophotometer (Agilent, Santa Clara, CA, USA) at wavelength 436 nm at CEH UK Centralised Analytical Chemistry facility, Lancaster. Additionally, soil moisture, in the form of volumetric water content, was measured at 7 and 18 cm depths at the same location using CS616 soil moisture probes (Campbell Scientific). WTD was measured using a Level TROLL 400 Data Logger (In-Situ, Fort Collins, CO, USA). Automated WTD measurements were highly correlated with values from five nearby manual dip wells ($R^2 = 0.94$), indicating good representativeness of the measurements for the area. All automated measurements were recorded hourly.

2.5. Methane Production and Oxidation Potential

To determine the CH_4 production and oxidation potential in the peat profile during the transition from nongrowing to growing-season, laboratory incubations of peat cores collected monthly from March–September 2016 were performed: Three 100 cm cores were collected at each sampling event using a Russian peat corer. To reduce damage to the peat from repeated sampling, each core was collected at 1 m distance from a central point and the sampling position rotated by 45° at each sampling event (Figure S1 in Supporting Information S1). Peat cores were divided into 10 cm segments, placed into self-sealed bags, transported to the laboratory and stored at 4°C until incubation. Incubations were started within 5 days of sampling. Incubation experiments were conducted on core segments at seven depth intervals (0–10 cm, 10–20 cm, 20–30 cm, 30–40 cm, 50–60 cm, 70–80 cm, and 90–100 cm). Laboratory incubations to determine CH_4 production and oxidation potential were adapted from Sundh et al. (1995) and Kettunen et al. (1999). In brief, peat samples were homogenized and 25 g aliquots were placed into 250 mL Kilner jars with septa in the lids and amended with 25 mL de-ionized water to achieve a slurry. Two sets of samples were prepared for CH_4 production and oxidation potential. For CH_4 production incubations, jars were flushed with N_2 gas for 120 s and then immediately sealed and injected with 40 mL N_2 gas to obtain excess pressure to compensate for gas removed during headspace sampling. For CH_4 oxidation, incubation samples were treated similarly, but synthetic compressed air (20.69% O_2 , <1 ppm CO_2 , <2 ppm H_2O , <0.1 ppm NO_x , <0.1 ppm total hydrocarbons) was used instead of N_2 gas. In addition, oxidation samples were injected with 5 mL 100,000 ppm CH_4 gas (CK Special gases LTD, Leicestershire, UK) to obtain an initial concentration of around 1,600 ppm CH_4 . During incubation, samples were stored on a rotary shaker at 20°C in the dark. This temperature was chosen as it is close to the optimum for soil microorganisms (Pietikäinen et al., 2005) to ensure sufficient microbial activity and provide consistent conditions across all incubations. To assess the rate of change of CH_4 concentration, 5 mL headspace samples were collected using a syringe at 4 timepoints: the start of the incubation and then after 1, 3, and 7 days. Samples were transferred into evacuated 3 mL Exetainer vials and CH_4 concentrations were measured by gas chromatography as described above. Production and oxidation rates were calculated from linear regressions as described by Sundh et al. (1995). Regressions of CO_2 production rates were used for quality control and regressions with $R^2 < 0.9$ excluded from the analysis. Potential CH_4 production and oxidation rates were normalized to sample dry weight.

2.6. Statistical Analysis

Mean flux rates between chamber locations varied by up to three orders of magnitude. For visualization and computing correlations with environmental variables, flux rates were transformed using the inverse hyperbolic sine (ihs), defined as $\sin^{-1} x = \log(x + \sqrt{x^2 + 1})$. This is an alternative to taking the logarithm, which allows for the inclusion of zero and negative values (Burbidge et al., 1988; Johnson, 1949). Soil solution chemistry data measured every 2 weeks was interpolated to obtain weekly values using Stineman interpolation (Stineman, 1980). Similarly, gaps in soil moisture data due to instrument failure of up to 6 hr were interpolated. To test for delays in the response of GHG fluxes to environmental conditions, we used cross-correlation with a maximum time lag of 14 days. All environmental variables correlated most strongly with fluxes on the day of measurement; thus, fluxes

were matched with environmental variables at the time of measurement. To determine the association between fluxes and environmental variables at the chamber level, we used repeated measures correlation (rmcorr) (Bakdash & Marusich, 2017; Bland & Altman, 1995). Rmcorr accounts for nonindependence among cases (in this study the flux chamber locations) with repeated observations. It provides a best fit regression line for each chamber with a common slope but varying intercepts. Analogous to a Pearson correlation coefficient, the rmcorr coefficient (r_{rm}) is bounded between -1 and 1 and represents the strength of the association between two variables (Bakdash & Marusich, 2017). Pearson correlation coefficients were calculated for variables measured without spatial replication. For the analysis of CH_4 production and oxidation potentials, one-way ANOVA followed by Tukey's HSD test was used. Where necessary, data were $\ln$ -transformed to meet the assumptions of ANOVA.

Stable isotope values of CH_4 were determined from the five samples collected at each time point. Sample CH_4 concentrations above atmospheric background ranged from ~ 0.7 to 311 ppm (median 4 ppm). To calculate the emission source isotope values from the sampled mix of emitted and atmospheric background CH_4 , we used Miller-Tans plots: the product of CH_4 concentration and isotope δ -value are plotted against CH_4 concentration and the source isotope signature is given by the slope of a regression (Miller & Tans, 2003). For the regression, we used York's method to obtain unbiased estimates of the regression parameters that allow for errors in both variables and produce an uncertainty estimate that is based on the empirical instrument precision (Wehr & Saleska, 2017; York, 1968). Uncertainty is given as the standard error of the regression slope.

All statistical analysis and data visualization were performed in R version 4.5.1 (R Core Team, 2025) with the additional use of the rmcorr (Bakdash & Marusich, 2024), IsoplotR (Vermeesch, 2018), multcomp (Hothorn et al., 2008), imputeTS (Moritz & Bartz-Beielstein, 2017), scipub (Pagliaccio, 2024), patchwork (Pedersen, 2025), ggpubr (Kassambara, 2025), ggkabeito (Barrett, 2021), scales (Wickham et al., 2025) and tidyverse (Wickham et al., 2019) packages.

3. Results

3.1. CH_4 Fluxes, Ecosystem Respiration, and Environmental Factors

Flux rates of CH_4 showed substantial variation between chamber locations and across seasons, with values ranging from $-0.6 \mu\text{mol m}^{-2} \text{h}^{-1}$ in March 2016 to $45,000 \mu\text{mol m}^{-2} \text{h}^{-1}$ in the summer of 2016 (Figure 1a). Median fluxes ranged from 10.6 to $91.8 \mu\text{mol m}^{-2} \text{h}^{-1}$. Peak fluxes in all chamber locations occurred between June and November each year; notably higher fluxes were observed in 2016 with flux rates in all locations reaching values 3 to 8 standard deviations above their respective means in that year alone (Figure S3 in Supporting Information S1).

Ecosystem respiration (R_{eco}), that is, dark chamber CO_2 emissions, ranged from $6 \mu\text{mol m}^{-2} \text{h}^{-1}$ to $23,900 \mu\text{mol m}^{-2} \text{h}^{-1}$, with median values of $1,520 \mu\text{mol m}^{-2} \text{h}^{-1}$ to $2,560 \mu\text{mol m}^{-2} \text{h}^{-1}$, and followed a similar seasonal pattern as CH_4 (Figure S4 in Supporting Information S1). CO_2 and CH_4 fluxes were significantly correlated with each other (Figure 2).

WTD was consistently high, ranging from -22 cm to $+7$ cm relative to the peat surface with a mean annual depth of -2 cm over the three years covered by this study (Figure 1c). Similarly, soil moisture at 18 cm depth was consistently high ranging from 91% in September 2016 to 97% in February 2016 with a mean of 95% (Figure S5 in Supporting Information S1). Mean air temperatures for the years 2015, 2016, and 2017 were 5.8, 6.1, and 6.4°C , respectively. However, air and soil temperatures were highest during the summer period of 2016, coinciding with the highest CH_4 and CO_2 fluxes (Figure 1b, Figure S6 in Supporting Information S1). Across all chamber locations, there was a positive nonlinear relationship between both CH_4 and CO_2 fluxes and soil temperature at 30 cm depth (Figure 2a), whereby $\ln(CH_4)$ increased by 0.360 for every 1°C increase. Water tables showed a unimodal relationship with CH_4 fluxes: highest fluxes were measured between WTD of -7 cm to $+3$ cm and decreased at both lower and higher values (Figure S7 in Supporting Information S1). Exploratory analysis revealed that soil solution color showed a positive correlation with CH_4 flux and was highly correlated with soil temperature, while soil moisture showed a negative correlation and was also highly negatively correlated with soil temperature (Table 1). Soil variables measured at multiple depths showed similar patterns. Temperature at 30 cm and soil solution chemistry at 10 cm were the strongest predictors of CH_4 fluxes, while CO_2 fluxes were most strongly correlated with temperatures at 10 and 30 cm depth (Table S2 in Supporting Information S1).

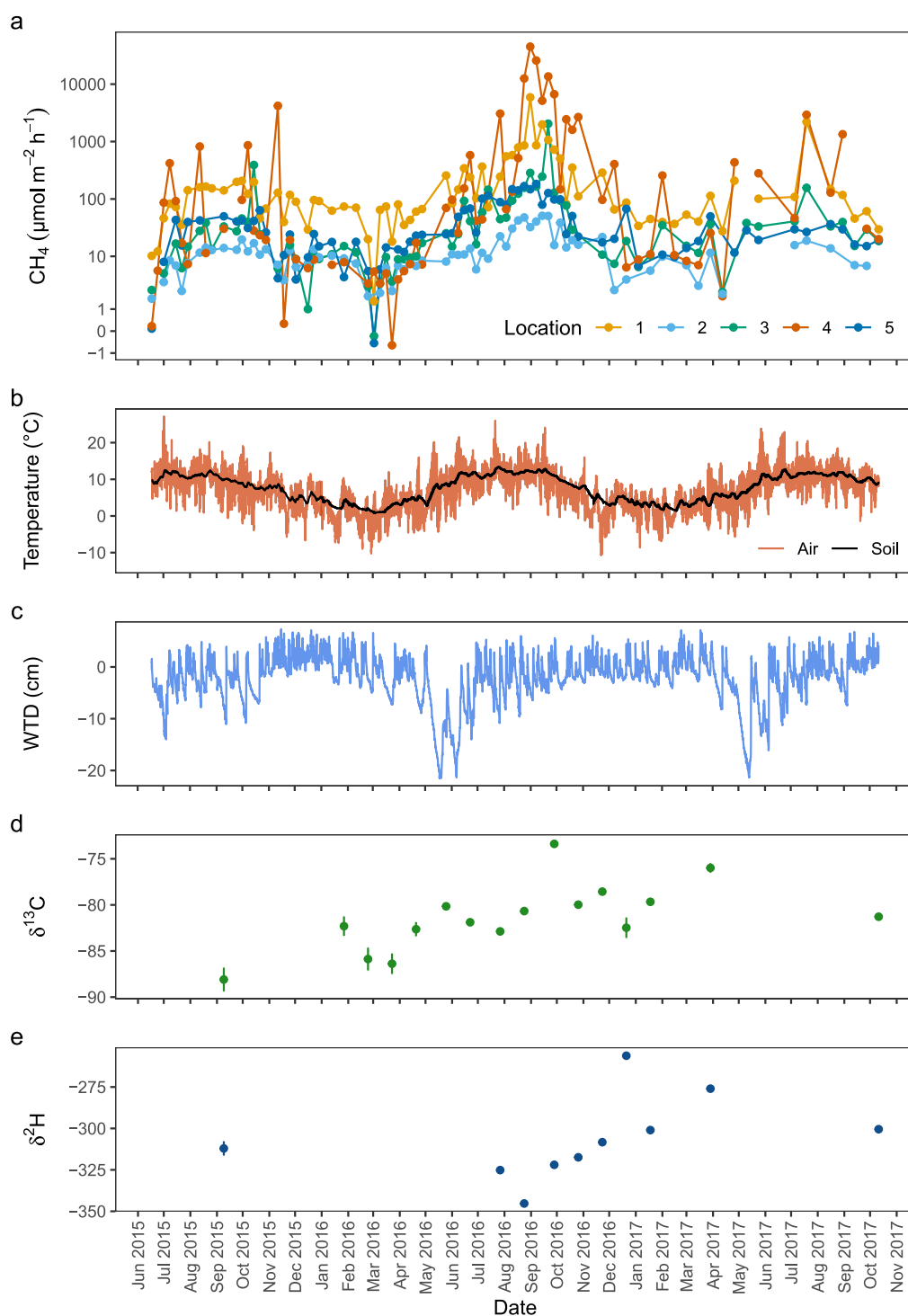


Figure 1. Time series of (a) CH₄ flux at five chamber locations, (b) air temperature and soil temperature at 30 cm depth, (c) WTD with positive values indicates water levels above the soil surface, (d-e) δ¹³C and δ²H of emitted CH₄ ± SE.

3.2. Methane Isotope Ratios

Both values for δ¹³C and δ²H varied across time and ranged from −88.4‰ to −73.2‰ and −345.3‰ to −275.6‰, respectively (Figures 1d and 1e). Notably, δ¹³C and δ²H showed opposing trends with respect to the

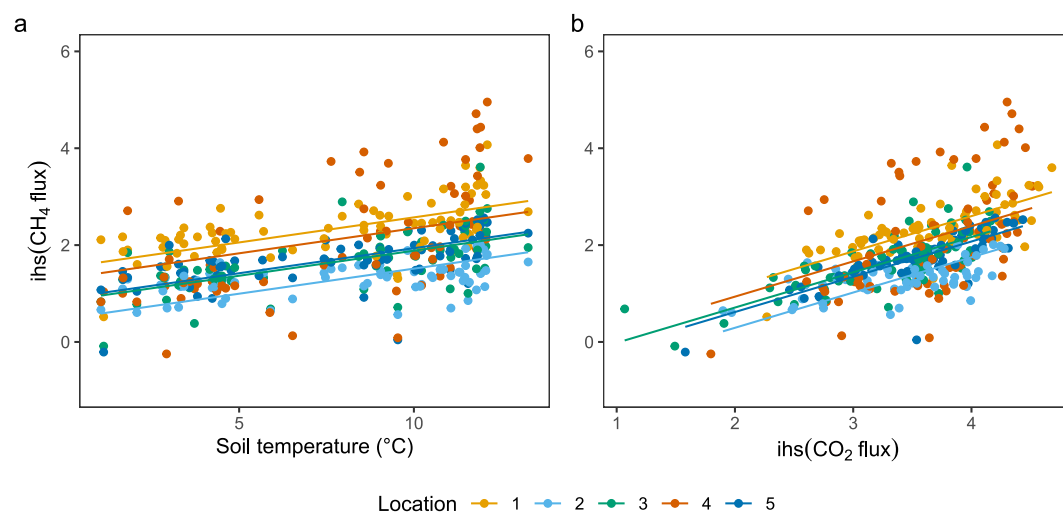


Figure 2. Repeated measures correlation between CH_4 flux and (a) soil temperature at 30 cm depth ($r_{rm} = 0.54$, $p < 0.001$) (b) R_{eco} , that is, dark chamber CO_2 flux ($r_{rm} = 0.59$, $p < 0.001$). Fluxes of CH_4 and CO_2 were transformed using inverse hyperbolic sine (ihs) transformation.

CH_4 flux rate: $\delta^{13}\text{C}$ increased with CH_4 flux, leading to generally higher values in summer/autumn, while $\delta^2\text{H}$ decreased with CH_4 flux, leading to generally higher values in winter/spring (Figure 3). There was no relationship between $\delta^{13}\text{C}$ and $\delta^2\text{H}$ (Figure S8 in Supporting Information S1).

Correlation analysis was performed for the subset of 16 samplings with isotope measurements (Table 2). Similar trends as in the whole data set were observed; $\delta^2\text{H}$ decreased significantly with soil temperature and soil solution color.

3.3. Methane Production and Oxidation Potentials

Potential production rates of CH_4 were variable, ranging from $-2.3 \text{ nmol CH}_4 \text{ m}^{-2} \text{ h}^{-1}$ to $369 \text{ nmol CH}_4 \text{ m}^{-2} \text{ h}^{-1}$ (Figure 4). Rates changed over the 7-month sampling period, with production rates in April significantly higher than those in other months, except for March and September (Tukey HSD, $p < 0.05$). When standardizing measurements across sampling times by calculating standard scores, production rates were seen to be significantly higher at 30–40 cm depth compared to other depths, except for 90–100 cm. Oxidation potentials ranged from $-3.3 \text{ CH}_4 \text{ m}^{-2} \text{ h}^{-1}$ to $2.3 \text{ CH}_4 \text{ m}^{-2} \text{ h}^{-1}$ and there was significantly less oxidation in September than in March as well as a significant decrease with depth in the peat profile (Figure S9 in Supporting Information S1).

Table 1
Correlations of GHG Fluxes and Environmental Parameters

	ihs(CO_2 flux)	Soil temperature	Water table	Soil moisture	Soil solution DOC	Soil solution color	pH
ihs(CH_4 flux)	0.59***	0.54***	n.a.	−0.57***	0.39***	0.61***	n.a.
ihs(CO_2 flux)		0.72***	n.a.	−0.62***	0.39***	0.61***	n.a.
Soil temperature			−0.24*	−0.90***	0.38***	0.74***	−0.26*
Water table				0.24	0.01	−0.05	0
Soil moisture					−0.48***	−0.77***	0.42**
Soil solution DOC						0.77***	−0.50***
Soil solution color							−0.63***

Note. Values are repeated measures correlation coefficients for correlations with ihs(CH_4 flux) and ihs(CO_2 flux) and Pearson correlation coefficients among other soil parameters. Soil temperature and soil moisture measured at 30 and 18 cm, respectively; soil solution DOC, soil solution color, and soil pH measured at 10 cm depth, respectively. See Table S2 in Supporting Information S1 for additional depths. Significance given as * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$; n.a. Indicates nonlinear relationships between variables.

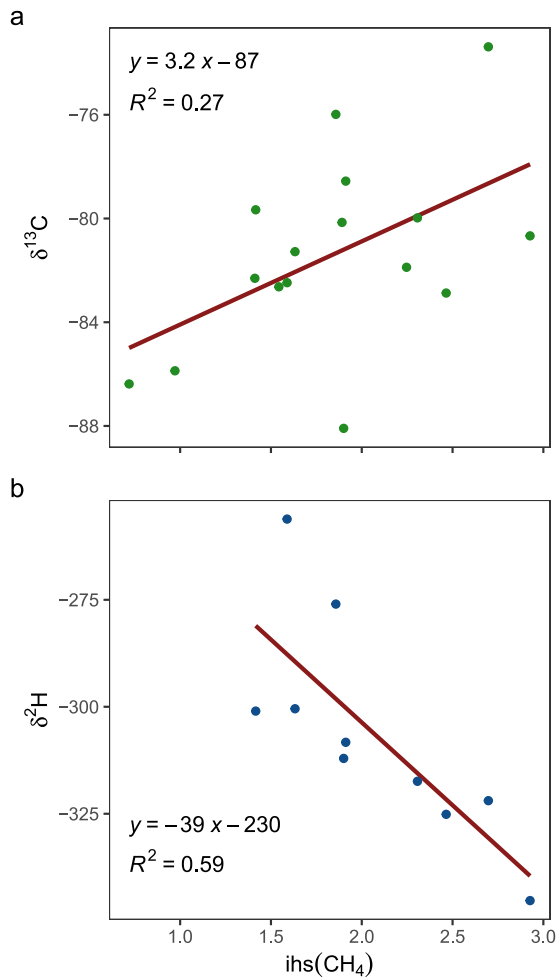


Figure 3. Regression of CH₄ isotopic value against ihs(CH₄) flux for (a) δ¹³C ($p = 0.04$) and (b) δ²H ($p = 0.009$).

and air temperatures. This is likely because CH₄ fluxes are largely driven by soil temperature below the WTD and there is a lag between changes in air and soil temperatures (Figure 1b), which increases with depth

4. Discussion

Seasonal and interannual variations in peatland CH₄ emissions arise from changes in methanogenic and methanotrophic activity. Stable isotopes are key tracers of CH₄ production and oxidation processes. Here, we used a combination of CH₄ flux and dual stable isotope (δ¹³C-CH₄, δ²H-CH₄) measurements as well as laboratory incubations and environmental data collected over a 28-month period to investigate seasonal variation in CH₄ biogeochemistry of a temperate peatland. We hypothesized that as temperature and water table altered the relative activity of aerobic and anaerobic microbial activity in the peat, CH₄ fluxes and their isotopic ratios would co-vary across seasons.

4.1. Fluxes and Environmental Drivers

Net emissions of CH₄ varied strongly between seasons and years. Our sampling covered a period of unusually high fluxes in 2016 with peak emissions around one order of magnitude above those of the years 2015 and 2017. Values of up to 45,000 μmol m⁻² h⁻¹ were recorded, exceeding maximum values found in a large data set for temperate and boreal peatlands (Turetsky et al., 2014) and considerably higher than what has previously been found in UK peatlands (Levy et al., 2012). Unusually high CH₄ fluxes have also been found in other peatlands in northern England between 2015 and 2017 (Heinemeyer, 2023) as well as in the wider boreal-arctic region in 2016 due to anomalously high temperatures (Yuan et al., 2024). As expected, fluxes were related to temperature, which is an established driver of wetland CH₄ emissions (Yvon-Durocher et al., 2014). Previous studies on the same peatland have found increases in emissions in response to increases in air and soil temperatures (Ward et al., 2007, 2013), as well as a stronger positive temperature response of methanogens compared to methanotrophs (Windén et al., 2012). Temperatures at our site also exceeded the long-term average during the late summer in 2016 (Figure S6 in Supporting Information S1) and were likely an important driver behind the high observed CH₄ fluxes, while other factors may have contributed to the high emissions. The annual period of highest CH₄ emissions extends from around June to November each year and is therefore delayed relative to the annual summer or growing season

Table 2

Correlation Coefficients of CH₄ Stable Isotope Values, Mean CH₄ Fluxes, and Soil Parameters for the Subset of Samplings Where Stable Isotope Samples Were Collected

	δ ² H	ihs(CH ₄ flux)	Soil temperature	Water table	Soil moisture	Soil solution DOC	Soil solution color	pH
δ ¹³ C	0.01	0.52*	0.11	0.23	-0.1	0.04	0.16	-0.06
δ ² H		-0.77**	-0.73*	0.43	0.67	0	-0.75*	0.15
ihs(CH ₄ flux)			0.82***	0.01	-0.87***	0.72**	0.88***	-0.51*
Soil temperature				-0.18	-0.85***	0.69**	0.89***	-0.36
Water table					0.17	-0.21	-0.18	-0.02
Soil moisture						-0.73**	-0.90***	0.74**
Soil solution DOC							0.89***	-0.52*
Soil solution color								-0.57*

Note. * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$; n.a. indicates nonlinear relationships between variables.

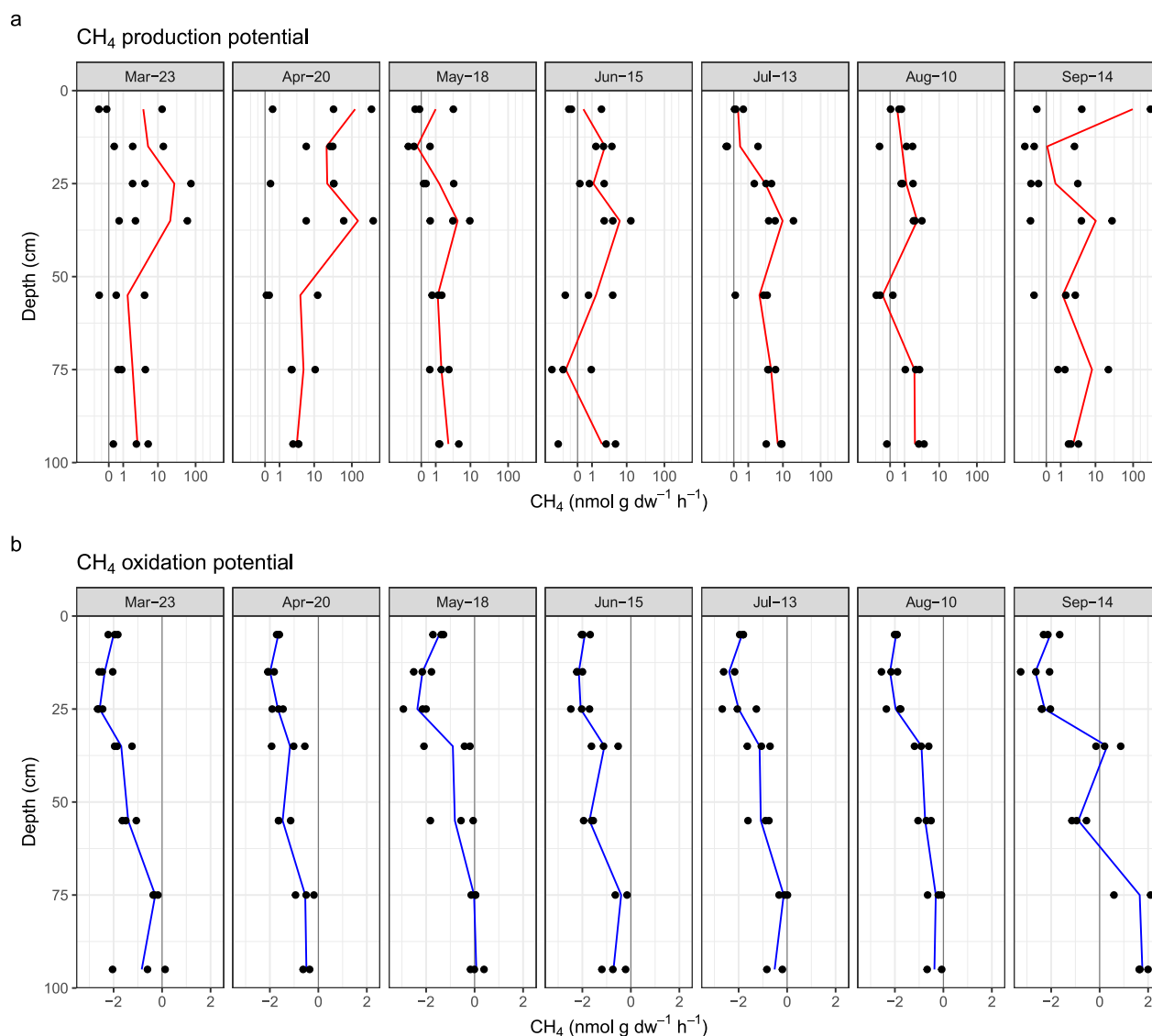


Figure 4. Potential CH_4 (a) production and (b) oxidation within the peat profile of three replicate peat cores; mean values are connected by continuous lines. Sampling dates of peat cores during the 2016 growing season are shown at the top of panels. Note that rates for (a) are plotted on an inverse hyperbolic sine scale to visualize a large range of values. Negative values indicate CH_4 removal, positive values CH_4 release.

(Dawson & Fisher, 1964). Soil temperatures at 30 cm depth, the zone of highest methane production potential, were permissive of CH_4 production year-round and would only briefly drop below 4°C in winter.

WTD is a key driver of CH_4 fluxes in peatlands (Evans et al., 2021; Moore & Roulet, 1993), though its effect depends on local conditions at the plot-scale (Dinsmore et al., 2009). However, variations in WTD during this study were <30 cm and showed limited seasonal variation. Thus, conditions conducive to methanogenesis and limited methane oxidation were maintained throughout the year (Figure 1c). This is supported by historic data from the same catchment showing high water tables either above or within 5 cm of the mire surface 83% of the time (Evans et al., 1999). While CH_4 emissions are generally expected to increase with rising water table, we observed relatively lower fluxes during both times of lower water table and inundation (Figure S7 in Supporting Information S1), similar to results reported for bogs by Turetsky et al. (2014). A reduction in emissions during inundation may be due to a decrease in substrate availability as a result of reduced plant inputs as well as soil temperature (Strack et al., 2004). The positive relationship between CO_2 and CH_4 emissions also indicates that the system did not experience prolonged periods of oxidizing conditions characterized by low CH_4 and high CO_2 fluxes (Figure 2b).

Emissions were also related to water chemistry (Table 1). DOC provides substrate for microorganisms and has been previously shown to positively correlate with both CH₄ and CO₂ emissions (Liang et al., 2024; Pastor et al., 2003; Zhang et al., 2022). The correlation between CH₄ flux and soil solution color is more ambiguous, as increases in soil solution color, resulting from the release of tannins during decomposition, possibly integrate factors such as temperature, decomposer activity, and hydrology, rather than being a direct cause of increased methanogenesis. However, the optical properties of soil solution in peatlands can serve as a useful proxy of ecosystem processes (Hansen et al., 2016) and Liang et al. (2024) have found that CH₄ emissions in a saltmarsh increased with the humification index.

There was a high degree of variability in CH₄ fluxes between chamber locations, particularly during the summer and autumn where fluxes varied by up to two orders of magnitude (Figure 1a, Figure S10 in Supporting Information S1). Part of this variability may be due to the differences in microtopography, microclimate, and vegetation between individual locations, which were not captured by our analysis. For instance, the locations with the highest CH₄ emissions had the highest abundance of graminoids (Figure S2, Table S1 in Supporting Information S1), which are known to positively correlate with CH₄ fluxes (Gray et al., 2013; Limpens et al., 2008; Ward et al., 2013).

4.2. Methane Production and Oxidation Potentials

The laboratory peat incubations revealed no clear seasonal pattern in CH₄ production and oxidation potentials, although the highest production rates were observed in April (Figure 4). However, across all incubations, production potentials were highest between 30 and 40 cm depth, while oxidation potentials decreased below the water table (Figure S9 in Supporting Information S1). This is consistent with previous research showing the highest CH₄ production potential just below the average WTD (Sundh et al., 1994). Care must be taken when extrapolating from such incubation experiments to field conditions as only potential rates are measured. In some cases, net fluxes showed the opposite sign expected during anoxic or oxic conditions. Potential explanations for this include, for example, anaerobic CH₄ oxidation or, conversely, release of residual CH₄ from the sample. However, our results indicate that WTD remained consistently above the main zone of methanogenesis in the peat profile and that the maximum CH₄ production potential in the peat column was substantially higher than the maximum CH₄ oxidation potential throughout the measured period. This suggests that conditions at the study site favored methanogenesis throughout the season with relatively limited potential for CH₄ oxidation and associated stable isotope fractionation (Sundh et al., 1994, 1995).

4.3. Methane Isotope Ratios

We expected to find seasonal variations in the stable isotope composition of CH₄ primarily in response to changes in the relative magnitude of CH₄ production and oxidation processes. Stable isotope composition is the result of biological and physical processes interacting during CH₄ formation and release: fractionation during acetoclastic or hydrogenotrophic methanogenesis leads to a pathway-specific pattern of depletion in $\delta^{13}\text{C}$ and $\delta^2\text{H}$; fractionation during microbial oxidation leads to an enrichment in isotope signatures; transport through aerenchyma leads to a depletion in isotope signatures. The balance between production and oxidation in particular can be affected by factors such as temperature, substrate availability, and seasonal changes in plant mediated pathways (Le Mer & Roger, 2001). We found that while both $\delta^{13}\text{C}$ and $\delta^2\text{H}$ -values varied over time, they showed opposing trends: $\delta^2\text{H}$ had a strong negative relationship with CH₄ emission flux, while $\delta^{13}\text{C}$ showed an increase with CH₄ flux (Figure 3). Fractionation during CH₄ oxidation or transport is expected to shift isotope values in the same direction. However, changes in methanogenic pathways have opposing effects on C and H isotopes: acetoclastic methanogenesis is associated with a relative enrichment in ^{13}C but a relative depletion in ^2H , compared to hydrogenotrophic methanogenesis (Chanton et al., 2005). Therefore, our results indicate that increases in CH₄ emissions are primarily driven by acetoclastic methanogenesis. Similar observations have been made in other ecosystems. In both subtropical marine (Burke et al., 1988) as well as freshwater sediments (Burke et al., 1992), a negative relationship was observed between $\delta^{13}\text{C}$ and $\delta^2\text{H}$ of CH₄ with the highest $\delta^2\text{H}$ values observed during the colder months of the year. In an ombrotrophic peat bog, a shift from hydrogenotrophic methanogenesis in May to acetoclastic methanogenesis in July and September was found by Keller and Bridgman (2007), similar to what is suggested by the opposing relationships of $\delta^{13}\text{C}$ and $\delta^2\text{H}$ with CH₄ flux in this study. Such seasonal changes in metabolic pathways are likely the result of changes in substrate availability as well as environmental conditions. Acetoclastic methanogenesis is driven by plant inputs of labile C during the growing season, and senescent plant

tissues after the growing season (Galand et al., 2005; Sun et al., 2012). Hydrogenotrophic methanogens become relatively more prominent when there is a lack of plant inputs and during colder winter months (Sun et al., 2012).

It is acknowledged that peatlands are heterogeneous ecosystems and stable isotope values can vary not only over time but also with location. Measurements from a set of flux chambers may therefore not be representative of the entire ecosystem (Fisher et al., 2017). Furthermore, CH₄ production in peatlands is considered to be predominantly the result of acetoclastic methanogenesis (Kotsyurbenko et al., 2004), with the proportion of hydrogenotrophic methanogenesis increasing with peat depth (Chanton et al., 2005; Popp et al., 1999). There is evidence that methylotrophic methanogenesis may also occur in temperate peatlands, though this is rarely considered (Zalman et al., 2018). Methylotrophic methanogenesis results in strong ¹³C fractionation and comparatively low δ¹³C values (Penger et al., 2012). Our results are therefore still consistent with an increase in emissions driven by acetoclastic methanogenesis.

4.4. Implications of Seasonal Variability

Wetland stable isotope signatures follow regional and latitudinal patterns (Feinberg et al., 2018; Oh et al., 2022) and can vary on smaller scales between wetland types (Hornibrook & Bowes, 2007; Takriti et al., 2023). Time series of CH₄ stable isotope values from wetland sites are relatively rare (e.g., McCalley et al., 2014), and single measurements should be considered as a snapshot of conditions at a specific point in time. Temporal changes in isotope values of CH₄ emissions due to shifts in the fraction of CH₄ oxidized are included in some biogeochemical models, for example, Oh et al. (2022). However, there is currently insufficient knowledge on how temporal shifts in metabolic pathways affect the isotopic signature of wetland emissions. Our study, which found temporal variations of 15‰ δ¹³C and 89‰ δ²H from a single peatland site, therefore points toward a critical gap in our understanding of CH₄ biogeochemistry. Recent research reiterates the importance of stable isotopes in constraining CH₄ budgets (Basu et al., 2022; Oh et al., 2026), which requires accurate representation of both spatial and temporal variability. While inferring broader patterns is beyond the scope of this study, together with findings from other systems (Burke et al., 1988, 1992), it suggests a need for future research to investigate the prevalence and underlying drivers of such variability in wetland ecosystems.

5. Conclusions

We have found CH₄ emissions in a temperate ombrotrophic peat bog to be highly variable and strongly driven by soil temperature, with unusually high fluxes in the summer and autumn of 2016. Such extreme events are difficult to capture in short term or infrequent sampling campaigns, but understanding their causes may be key for predicting the response of peatlands to climate change as well as the large interannual variability observed in global peatland emissions. Long-term measurement campaigns that can capture such rare events are therefore critical for understanding what are likely to be nonlinear responses in peatland ecosystems and how they may relate to climate change. We found that the isotopic values of CH₄ emissions vary in opposite directions for δ¹³C and δ²H with total flux. This suggests that in the absence of strong variation in WTD, changes in isotope values are due to shifts in metabolic pathways where emission rate increases are driven by a higher proportion of acetoclastic methanogenesis. Our results demonstrate the value of dual-isotope measurements in investigating CH₄ biogeochemical processes, as changes in the more commonly used δ¹³C measurements alone may be attributed solely to changes in CH₄ oxidation. Our results further have important implications for atmospheric inversion models that use stable isotopes to constrain CH₄ emission sources as well as for understanding biogeochemical processes as isotope signatures from peatlands are often assumed to be constant or change primarily in response to changes in WTD. However, if temporal patterns such as those in this study are confirmed in wetland systems more broadly, then this information may improve the ability of biogeochemical models to constrain CH₄ emissions at regional or global scales.

Conflict of Interest

The authors declare no conflicts of interest relevant to this study.

Availability Statement

Data from the UK ECN are available from <https://ecn.ac.uk>. We used two ECN data sets: meteorology (<https://doi.org/10.5285/fc9bcd1c-e3fc-4c5a-b569-2fe62d40f2f5>, Rennie et al., 2017a) and soil solution chemistry (<https://doi.org/10.5285/b330d395-68f2-47f1-8d59-3291dc02923b>, Rennie et al., 2017b). All other data used in this study as well as code for analysis and visualization are published on Zenodo. Data are available at <https://doi.org/10.5281/zenodo.18391367> (Takriti et al., 2026); R code is available at <https://doi.org/10.5281/zenodo.18392614> (Takriti, 2026).

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References

- Bakdash, J. Z., & Marusich, L. R. (2017). Repeated measures correlation. *Frontiers in Psychology*, 8, 456. <https://doi.org/10.3389/fpsyg.2017.00456>
- Bakdash, J. Z., & Marusich, L. R. (2024). Rmccor: Repeated Measures Correlation (Version 0.7.0). Retrieved from <https://CRAN.R-project.org/package=rmccor>
- Barrett, M. (2021). Ggokabeito: “Okabe-Ito” Scales for “ggplot2” and “ggraph”. <https://doi.org/10.32614/CRAN.package.ggokabeito>
- Basiliko, N., Blodau, C., Roehm, C., Bengtson, P., & Moore, T. R. (2007). Regulation of decomposition and methane dynamics across natural, commercially mined, and restored northern peatlands. *Ecosystems*, 10(7), 1148–1165. <https://doi.org/10.1007/s10021-007-9083-2>
- Basu, S., Lan, X., Dlugokencky, E., Michel, S., Schwietzke, S., Miller, J. B., et al. (2022). Estimating emissions of methane consistent with atmospheric measurements of methane and $\delta^{13}\text{C}$ of methane. *Atmospheric Chemistry and Physics*, 22(23), 15351–15377. <https://doi.org/10.5194/acp-22-15351-2022>
- Bland, J. M., & Altman, D. G. (1995). Calculating correlation coefficients with repeated observations: Part 1—Correlation within subjects. *BMJ British Medical Journal*, 310(6977), 446. <https://doi.org/10.1136/bmj.310.6977.446>
- Bridgman, S. D., Cadillo-Quiroz, H., Keller, J. K., & Zhuang, Q. (2013). Methane emissions from wetlands: Biogeochemical, microbial, and modeling perspectives from local to global scales. *Global Change Biology*, 19(5), 1325–1346. <https://doi.org/10.1111/gcb.12131>
- Burbidge, J. B., Magee, L., & Robb, A. L. (1988). Alternative transformations to handle extreme values of the dependent variable. *Journal of the American Statistical Association*, 83(401), 123–127. <https://doi.org/10.1080/01621459.1988.10478575>
- Burke, R. A., Jr., Barber, T. R., & Sackett, W. M. (1992). Seasonal variations of stable hydrogen and carbon isotope ratios of methane in sub-tropical freshwater sediments. *Global Biogeochemical Cycles*, 6(2), 125–138. <https://doi.org/10.1029/92GB00155>
- Burke, R. A., Jr., Martens, C. S., & Sackett, W. M. (1988). Seasonal variations of D/H and $^{13}\text{C}/^{12}\text{C}$ ratios of microbial methane in surface sediments. *Nature*, 332(6167), 829–831. <https://doi.org/10.1038/332829a0>
- Case, S. D. C., McNamara, N. P., Reay, D. S., & Whitaker, J. (2014). Can biochar reduce soil greenhouse gas emissions from a miscanthus bioenergy crop? *GCB Bioenergy*, 6(1), 76–89. <https://doi.org/10.1111/gcbb.12052>
- Chanton, J. P., Chaser, L., Glasser, P., & Siegel, D. (2005). Carbon and hydrogen isotopic effects on microbial methane from terrestrial environments. In *Stable isotopes and biosphere atmosphere interactions* (pp. 85–105). Elsevier. <https://doi.org/10.1016/B978-012088447-6/50006-4>
- Chasar, L. (2000). Methane concentration and stable isotope distribution as evidence of rhizospheric processes: Comparison of a fen and bog in the glacial Lake Agassiz peatland complex. *Annals of Botany*, 86(3), 655–663. <https://doi.org/10.1006/anbo.2000.1172>
- Chowdhury, T. R., & Dick, R. P. (2013). Ecology of aerobic methanotrophs in controlling methane fluxes from wetlands. *Applied Soil Ecology*, 65(December 2012), 8–22. <https://doi.org/10.1016/j.apsoil.2012.12.014>
- Ciais, P., Zhu, Y., Cai, Y., Lan, X., Michel, S. E., Zheng, B., et al. (2026). Why methane surged in the atmosphere during the early 2020s. *Science*, 391(6785), eadx8262. <https://doi.org/10.1126/science.adx8262>
- Dawson, G. B., & Fisher, R. G. (1964). Diurnal and seasonal ground temperature variations at wairakei. *New Zealand Journal of Geology and Geophysics*, 7(1), 144–154. <https://doi.org/10.1080/00288306.1964.10420166>
- Dean, J. F., Middelburg, J. J., Röckmann, T., Aerts, R., Blauw, L. G., Egger, M., et al. (2018). Methane feedbacks to the global climate system in a warmer world. *Reviews of Geophysics*, 56(1), 207–250. <https://doi.org/10.1002/2017RG000559>
- Deng, J., McCalley, C. K., Frolking, S., Chanton, J., Crill, P., Varner, R., et al. (2017). Adding stable carbon isotopes improves model representation of the role of microbial communities in peatland methane cycling. *Journal of Advances in Modeling Earth Systems*, 9(2), 1412–1430. <https://doi.org/10.1002/2016MS000817>
- Dinsmore, K. J., Skiba, U. M., Billett, M. F., Rees, R. M., & Drewer, J. (2009). Spatial and temporal variability in CH_4 and N_2O fluxes from a Scottish ombrotrophic peatland: Implications for modelling and up-scaling. *Soil Biology and Biochemistry*, 41(6), 1315–1323. <https://doi.org/10.1016/j.soilbio.2009.03.022>
- Dlugokencky, E. J., Nisbet, E. G., Fisher, R., & Lowry, D. (2011). Global atmospheric methane: Budget, changes and dangers. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*, 369(1943), 2058–2072. <https://doi.org/10.1098/rsta.2010.0341>
- Drinkwater, A., Palmer, P. I., Feng, L., Arnold, T., Lan, X., Michel, S. E., et al. (2023). Atmospheric data support a multi-decadal shift in the global methane budget towards natural tropical emissions. *Atmospheric Chemistry and Physics*, 23(14), 8429–8452. <https://doi.org/10.5194/acp-23-8429-2023>
- Evans, C., Morrison, R., Burden, A., Williamson, J., Baird, A., Brown, E., et al. (2017). *Final report on project SP1210: Lowland peatland systems in England and Wales – Evaluating greenhouse gas fluxes and carbon balances* (pp. 1–170). DEFRA. Retrieved from <http://randd.defra.gov.uk/Default.aspx?Module=More&Location=None&ProjectID=17584>
- Evans, C., Peacock, M., Baird, A. J., Artz, R. R. E., Burden, A., Callaghan, N., et al. (2021). Overriding water table control on managed peatland greenhouse gas emissions. *Nature*, 593(7860), 548–552. <https://doi.org/10.1038/s41586-021-03523-1>
- Evans, M. G., Burt, T. P., Holden, J., & Adamson, J. K. (1999). Runoff generation and water table fluctuations in blanket peat: Evidence from UK data spanning the dry summer of 1995. *Journal of Hydrology*, 221(3), 141–160. [https://doi.org/10.1016/S0022-1694\(99\)00085-2](https://doi.org/10.1016/S0022-1694(99)00085-2)
- Feinberg, A. I., Coulon, A., Stenke, A., Schwietzke, S., & Peter, T. (2018). Isotopic source signatures: Impact of regional variability on the $\delta^{13}\text{CH}_4$ trend and spatial distribution. *Atmospheric Environment*, 174(July 2017), 99–111. <https://doi.org/10.1016/j.atmosenv.2017.11.037>
- Fisher, R. E., France, J. L., Lowry, D., Lanoisellé, M., Brownlow, R., Pyle, J. A., et al. (2017). Measurement of the ^{13}C isotopic signature of methane emissions from northern European wetlands. *Global Biogeochemical Cycles*, 31(3), 605–623. <https://doi.org/10.1002/2016GB005504>

- Galand, P. E., Fritze, H., Conrad, R., & Yrjälä, K. (2005). Pathways for methanogenesis and diversity of methanogenic archaea in three boreal peatland ecosystems. *Applied and Environmental Microbiology*, 71(4), 2195–2198. <https://doi.org/10.1128/AEM.71.4.2195-2198.2005>
- Ganesan, A. L., Schwietzke, S., Poulter, B., Arnold, T., Lan, X., Rigby, M., et al. (2019). Advancing scientific understanding of the global methane budget in support of the paris agreement. *Global Biogeochemical Cycles*, 33(12), 1475–1512. <https://doi.org/10.1029/2018GB006065>
- Gray, A., Levy, P. E., Cooper, M. D. A., Jones, T., Gaiawyn, J., Leeson, S. R., et al. (2013). Methane indicator values for peatlands: A comparison of species and functional groups. *Global Change Biology*, 19(4), 1141–1150. <https://doi.org/10.1111/gcb.12120>
- Hansen, A. M., Kraus, T. E. C., Pellerin, B. A., Fleck, J. A., Downing, B. D., & Bergamaschi, B. A. (2016). Optical properties of dissolved organic matter (DOM): Effects of biological and photolytic degradation. *Limnology and Oceanography*, 61(3), 1015–1032. <https://doi.org/10.1002/lno.10270>
- Heinemeyer, A. (2023). *Restoration of heather-dominated blanket bog vegetation for biodiversity, carbon storage, greenhouse gas emissions and water regulation: Comparing burning to alternative mowing and uncut management: Final 10-year report to the Project Advisory Group of Peatland-ES-UK (Monograph)*. University of York. <https://doi.org/10.15124/yao-2wtg-kb53>
- Hornibrook, E. R. C., & Bowes, H. L. (2007). Trophic status impacts both the magnitude and stable carbon isotope composition of methane flux from peatlands. *Geophysical Research Letters*, 34(21), 2–6. <https://doi.org/10.1029/2007GL031231>
- Hothorn, T., Bretz, F., & Westfall, P. (2008). Simultaneous inference in general parametric models. *Biometrical Journal*, 50(3), 346–363. <https://doi.org/10.1002/bimj.200810425>
- Hurford, C., Rowell, T., Allen, D., & Brown, A. (2006). In C. Hurford & M. Schneider (Eds.), *Monitoring Nature conservation in cultural habitats*. Springer Netherlands. <https://doi.org/10.1007/1-4020-3757-0>
- Johnson, N. L. (1949). Systems of frequency curves generated by methods of translation. *Biometrika*, 36(1/2), 149–176. <https://doi.org/10.2307/232539>
- Kassambara, A. (2025). Ggpubr: ggplot2 based publication ready plots (Version 0.6.1). Retrieved from <https://CRAN.R-project.org/package=ggpubr>
- Keller, J. K., & Bridgman, S. D. (2007). Pathways of anaerobic carbon cycling across an ombrotrophic-minerotrophic peatland gradient. *Limnology and Oceanography*, 52(1), 96–107. <https://doi.org/10.4319/lno.2007.52.1.0096>
- Kettunen, A., Kaitala, V., Lehtinen, A., Lohila, A., Alm, J., Silvola, J., & Martikainen, P. J. (1999). Methane production and oxidation potentials in relation to water table fluctuations in two boreal mires. *Soil Biology and Biochemistry*, 31(12), 1741–1749. [https://doi.org/10.1016/S0038-0717\(99\)00093-0](https://doi.org/10.1016/S0038-0717(99)00093-0)
- Kirschke, S., Bousquet, P., Ciais, P., Saunio, M., Canadell, J. G., Dlugokencky, E. J., et al. (2013). Three decades of global methane sources and sinks. *Nature Geoscience*, 6(10), 813–823. <https://doi.org/10.1038/ngeo1955>
- Koffi, E. N., Bergamaschi, P., Alkama, R., & Cescatti, A. (2020). An observation-constrained assessment of the climate sensitivity and future trajectories of wetland methane emissions. *Science Advances*, 6(15), eaay4444. <https://doi.org/10.1126/sciadv.aay4444>
- Kotsyurbenko, O. R., Chin, K. J., Glagolev, M. V., Stubner, S., Simankova, M. V., Nozhevnikova, A. N., & Conrad, R. (2004). Acetoclastic and hydrogenotrophic methane production and methanogenic populations in an acidic West-Siberian peat bog. *Environmental Microbiology*, 6(11), 1159–1173. <https://doi.org/10.1111/j.1462-2920.2004.00634.x>
- Lai, D. Y. F. (2009). Methane dynamics in northern peatlands: A review. *Pedosphere*, 19(4), 409–421. [https://doi.org/10.1016/S1002-0160\(09\)0003-4](https://doi.org/10.1016/S1002-0160(09)0003-4)
- Lan, X., Nisbet, E. G., Dlugokencky, E. J., & Michel, S. E. (2021). What do we know about the global methane budget? Results from four decades of atmospheric CH₄ observations and the way forward. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences*, 379(2210), 20200440. <https://doi.org/10.1098/rsta.2020.0440>
- Le Mer, J., & Roger, P. (2001). Production, oxidation, emission and consumption of methane by soils: A review. *European Journal of Soil Biology*, 37(1), 25–50. [https://doi.org/10.1016/S1164-5563\(01\)01067-6](https://doi.org/10.1016/S1164-5563(01)01067-6)
- Levin, I., Bergamaschi, P., Dörr, H., & Trapp, D. (1993). Stable isotopic signature of methane from major sources in Germany. *Chemosphere*, 26(1–4), 161–177. [https://doi.org/10.1016/0045-6535\(93\)90419-6](https://doi.org/10.1016/0045-6535(93)90419-6)
- Levy, P. E., Burden, A., Cooper, M. D. A., Dinsmore, K. J., Drewer, J., Evans, C., et al. (2012). Methane emissions from soils: Synthesis and analysis of a large UK data set. *Global Change Biology*, 18(5), 1657–1669. <https://doi.org/10.1111/j.1365-2486.2011.02616.x>
- Liang, W., Chen, X., Chen, Z. L., Zhu, P., Huang, Z., Li, J., et al. (2024). Unraveling the impact of Spartina alterniflora invasion on greenhouse gas production and emissions in coastal saltmarshes: New insights from dissolved organic matter characteristics and surface-porewater interactions. *Water Research*, 262, 122120. <https://doi.org/10.1016/j.watres.2024.122120>
- Limpens, J., Berendse, F., Blodau, C., Canadell, J. G., Freeman, C., Holden, J., et al. (2008). Peatlands and the carbon cycle: From local processes to global implications - A synthesis. *Biogeosciences*, 5, 1475–1491. <https://doi.org/10.5194/bgd-5-1379-2008>
- Mannisenaho, V., Tsuruta, A., Backman, L., Houweling, S., Segers, A., Krol, M., et al. (2023). Global atmospheric δ¹³C_{CH₄} and CH₄ trends for 2000–2020 from the atmospheric transport model TM5 using CH₄ from carbon tracker Europe–CH₄ inversions. *Atmosphere*, 14(7), 1121. <https://doi.org/10.3390/atmos14071121>
- McCalley, C. K., Woodcroft, B. J., Hodgkins, S. B., Wehr, R. a., Kim, E.-H., Mondav, R., et al. (2014). Methane dynamics regulated by microbial community response to permafrost thaw. *Nature*, 514(7523), 478–481. <https://doi.org/10.1038/nature13798>
- McEwing, K. R., Fisher, J. P., & Zona, D. (2015). Environmental and vegetation controls on the spatial variability of CH₄ emission from wet-sedge and tussock tundra ecosystems in the arctic. *Plant and Soil*, 388(1–2), 37–52. <https://doi.org/10.1007/s11104-014-2377-1>
- McNamara, N. P., Plant, T., Oakley, S., Ward, S. E., Wood, C., & Ostle, N. (2008). Gully hotspot contribution to landscape methane (CH₄) and carbon dioxide (CO₂) fluxes in a northern peatland. *Science of the Total Environment*, 404(2–3), 354–360. <https://doi.org/10.1016/j.scitotenv.2008.03.015>
- Melton, J. R., Wania, R., Hodson, E. L., Poulter, B., Ringeval, B., Spahni, R., et al. (2013). Present state of global wetland extent and wetland methane modelling: Methodology of a model inter-comparison project (WETCHIMP). *Geoscientific Model Development*, 6(3), 617–641. <https://doi.org/10.5194/gmd-6-617-2013>
- Menoud, M., van der Veen, C., Lowry, D., Fernandez, J. M., Bakkaloglu, S., France, J. L., et al. (2022). New contributions of measurements in Europe to the global inventory of the stable isotopic composition of methane. *Earth System Science Data*, 14(9), 4365–4386. <https://doi.org/10.5194/essd-14-4365-2022>
- Miller, J. B., & Tans, P. P. (2003). Calculating isotopic fractionation from atmospheric measurements at various scales. *Tellus Series B Chemical and Physical Meteorology*, 55(2), 207–214. <https://doi.org/10.3402/tellusb.v55i2.16697>
- Moore, T. R., & Roulet, N. T. (1993). Methane flux: Water table relations in northern wetlands. *Geophysical Research Letters*, 20(7), 587–590. <https://doi.org/10.1029/93GL00208>
- Moritz, S., & Bartz-Beielstein, T. (2017). imputeTS: Time series missing value imputation in R. *The R Journal*, 9(1), 207–218. <https://doi.org/10.32614/RJ-2017-009>

- Oh, Y., Bruhwiler, L., Lan, X., Halder, S., Riddell-Young, B., Basu, S., et al. (2026). Isotopic constraints in methane inversions reveal larger trends in wetland emissions with improved linkage to terrestrial water storage. *Nature Communications*, 17(1), 8112. <https://doi.org/10.1038/s41467-026-74777-4>
- Oh, Y., Zhuang, Q., Welp, L. R., Liu, L., Lan, X., Basu, S., et al. (2022). Improved global wetland carbon isotopic signatures support post-2006 microbial methane emission increase. *Communications Earth & Environment*, 3(1), 1–12. <https://doi.org/10.1038/s43247-022-00488-5>
- Pagliaccio, D. (2024). Scipub: Summarize data for scientific publication (Version 1.2.3). Retrieved from <https://CRAN.R-project.org/package=scipub>
- Pastor, J., Solin, J., Bridgman, S. D., Updegraff, K., Harth, C., Weishampel, P., & Dewey, B. (2003). Global warming and the export of dissolved organic carbon from boreal peatlands. *Oikos*, 100(2), 380–386. <https://doi.org/10.1034/j.1600-0706.2003.11774.x>
- Pedersen, T. L. (2025). Patchwork: The Composer of Plots (Version 1.3.1). Retrieved from <https://CRAN.R-project.org/package=patchwork>
- Peng, S., Lin, X., Thompson, R. L., Xi, Y., Liu, G., Hauglustaine, D., et al. (2022). Wetland emission and atmospheric sink changes explain methane growth in 2020. *Nature*, 612(7940), 477–482. <https://doi.org/10.1038/s41586-022-05447-w>
- Penger, J., Conrad, R., & Blaser, M. (2012). Stable carbon isotope fractionation by methylophilic methanogenic archaea. *Applied and Environmental Microbiology*, 78(21), 7596–7602. <https://doi.org/10.1128/AEM.01773-12>
- Pietikäinen, J., Pettersson, M., & Bååth, E. (2005). Comparison of temperature effects on soil respiration and bacterial and fungal growth rates. *FEMS Microbiology Ecology*, 52(1), 49–58. <https://doi.org/10.1016/j.femsec.2004.10.002>
- Popp, T. J., Chanton, J. P., Whiting, G. J., & Grant, N. (1999). Methane stable isotope distribution at a carex dominated fen in north central Alberta. *Global Biogeochemical Cycles*, 13(4), 1063–1077. <https://doi.org/10.1029/1999GB900060>
- R Core Team. (2025). *R: A Language and environment for statistical computing*. R Foundation for Statistical Computing. Retrieved from <https://www.R-project.org/>
- Rennie, S., Adamson, J., Anderson, R., Andrews, C., Bater, J., Bayfield, N., et al. (2017a). UK Environmental Change Network (ECN) meteorology data: 1991–2015 [Dataset]. *NERC Environmental Information Data Centre*. <https://doi.org/10.5285/fc9bcd1c-e3fc-4c5a-b569-2fe62d40f2f5>
- Rennie, S., Adamson, J., Anderson, R., Andrews, C., Bater, J., Bayfield, N., et al. (2017b). UK Environmental Change Network (ECN) soil solution chemistry data: 1992–2015 [Dataset]. *NERC Environmental Information Data Centre*. <https://doi.org/10.5285/b330d395-68f2-47f1-8d59-3291dc02923b>
- Rennie, S., Andrews, C., Atkinson, S., Beaumont, D., Benham, S., Bowmaker, V., et al. (2020). The UK environmental change network datasets – Integrated and co-located data for long-term environmental research (1993–2015). *Earth System Science Data*, 12(1), 87–107. <https://doi.org/10.5194/essd-12-87-2020>
- Rodwell, J. S. (1993). *British plant communities, vol. 2, mires and heaths* (Vol. 81). Cambridge University Press. <https://doi.org/10.2307/2261244>
- Rose, R. J., Wood, C. M., Baxendale, C. L., Whitfield, M. G., Ostle, N. J., Driscoll, B., et al. (2016). Vegetation survey of Moor House National Nature reserve 2008–2009 [Text/csv Comma-separated values (CSV)]. <https://doi.org/10.5285/7A7D08E3-48E2-4AAD-855B-9D6767B9AE9B>
- Saunio, M., Martinez, A., Poulter, B., Zhang, Z., Raymond, P. A., Regnier, P., et al. (2025). Global methane budget 2000–2020. *Earth System Science Data*, 17(5), 1873–1958. <https://doi.org/10.5194/essd-17-1873-2025>
- Sherwood, O. A., Schwietzke, S., Arling, V. A., & Etiope, G. (2017). Global inventory of gas geochemistry data from fossil fuel, microbial and burning sources, version 2017. *Earth System Science Data*, 9(2), 639–656. <https://doi.org/10.5194/essd-9-639-2017>
- Smith, K. a., Ball, T., Conen, F., Dobbie, K. E., Massheder, J., & Rey, A. (2003). Exchange of greenhouse gases between soil and atmosphere: Interactions of soil physical factors and biological processes. *European Journal of Soil Science*, 54(December), 779–791. <https://doi.org/10.1046/j.1365-2389.2003.00567.x>
- Sriskantharajah, S., Fisher, R. E., Lowry, D., Aalto, T., Hatakka, J., Aurela, M., et al. (2012). Stable carbon isotope signatures of methane from a Finnish subarctic wetland. *Tellus B: Chemical and Physical Meteorology*, 64(1), 1–8. <https://doi.org/10.3402/tellusb.v64i0.18818>
- Stineman, R. W. (1980). A consistently well-behaved method of interpolation. *Creative Computing*, 6(7), 54–57.
- Strack, M., Waddington, J. M., & Tuittila, E.-S. (2004). Effect of water table drawdown on northern peatland methane dynamics: Implications for climate change. *Global Biogeochemical Cycles*, 18(4). <https://doi.org/10.1029/2003GB002209>
- Sun, C. L., Brauer, S. L., Cadillo-Quiroz, H., Zinder, S. H., & Yavitt, J. B. (2012). Seasonal changes in methanogenesis and methanogenic community in three peatlands, New York state. *Frontiers in Microbiology*, 3. <https://doi.org/10.3389/fmicb.2012.00081>
- Sundh, I., Mikkilä, C., Nilsson, M., & Svensson, B. H. (1995). Potential aerobic methane oxidation in a Sphagnum-dominated peatland-controlling factors and relation to methane emission. *Soil Biology and Biochemistry*, 27(6), 829–837. [https://doi.org/10.1016/0038-0717\(94\)0022-2-M](https://doi.org/10.1016/0038-0717(94)0022-2-M)
- Sundh, I., Nilsson, M., Granberg, G., & Svensson, B. H. (1994). Depth distribution of microbial production and oxidation of methane in northern boreal peatlands. *Microbial Ecology*, 27(3), 253–265. <https://doi.org/10.1007/BF00182409>
- Takriti, M. (2026). Moor House greenhouse gas flux and stable isotope analysis (Version v1.1.1) [Software]. *Zenodo*. <https://doi.org/10.5281/zenodo.22919681>
- Takriti, M., Chadwick, A., Rose, R. J., Dodd, B., Oakley, S., Wynn, P. M., et al. (2026). Moor House greenhouse gas fluxes and stable isotopes [Dataset]. *Zenodo*. <https://doi.org/10.5281/ZENODO.18391367>
- Takriti, M., Ward, S. E., Wynn, P. M., & McNamara, N. P. (2023). Isotopic characterisation and mobile detection of methane emissions in a heterogeneous UK landscape. *Atmospheric Environment*, 305, 119774. <https://doi.org/10.1016/j.atmosenv.2023.119774>
- Treat, C. C., Bloom, A. A., & Marushchak, M. E. (2018). Nongrowing season methane emissions—a significant component of annual emissions across northern ecosystems. *Global Change Biology*, 24(8), 3331–3343. <https://doi.org/10.1111/gcb.14137>
- Turetsky, M. R., Kotowska, A., Bubier, J., Dise, N. B., Crill, P., Hornibrook, E. R. C., et al. (2014). A synthesis of methane emissions from 71 northern, temperate, and subtropical wetlands. *Global Change Biology*, 20(7), 2183–2197. <https://doi.org/10.1111/gcb.12580>
- Turner, A. J., Frankenberg, C., Wennberg, P. O., & Jacob, D. J. (2017). Ambiguity in the causes for decadal trends in atmospheric methane and hydroxyl. *Proceedings of the National Academy of Sciences*, 114(21), 5367–5372. <https://doi.org/10.1073/pnas.1616020114>
- Umezawa, T., Machida, T., Aoki, S., & Nakazawa, T. (2012). Contributions of natural and anthropogenic sources to atmospheric methane variations over western Siberia estimated from its carbon and hydrogen isotopes. *Global Biogeochemical Cycles*, 26(4). <https://doi.org/10.1029/2011GB004232>
- Vermesch, P. (2018). IsoplotR: A free and open toolbox for geochronology. *Geoscience Frontiers*, 9(5), 1479–1493. <https://doi.org/10.1016/j.gsf.2018.04.001>
- Wahlen, M., Tanaka, N., Henry, R., Deck, B., Broeker, W., Shemesh, A., et al. (1990). ^{13}C , D , and ^{14}C in methane. In *NASA upper atmosphere research Program: Research summaries 1988–1989*.

- Ward, S. E., Bardgett, R. D., McNamara, N. P., Adamson, J. K., & Ostle, N. J. (2007). Long-term consequences of grazing and burning on northern peatland carbon dynamics. *Ecosystems*, 10(7), 1069–1083. <https://doi.org/10.1007/s10021-007-9080-5>
- Ward, S. E., Ostle, N. J., Oakley, S., Quirk, H., Henrys, P. a., & Bardgett, R. D. (2013). Warming effects on greenhouse gas fluxes in peatlands are modulated by vegetation composition. *Ecology Letters*, 16(10), 1285–1293. <https://doi.org/10.1111/ele.12167>
- Wehr, R., & Saleska, S. R. (2017). The long-solved problem of the best-fit straight line: Application to isotopic mixing lines. *Biogeosciences*, 14(1), 17–29. <https://doi.org/10.5194/bg-14-17-2017>
- Whiticar, M. J. (1999). Carbon and hydrogen isotope systematics of bacterial formation and oxidation of methane. *Chemical Geology*, 161(1–3), 291–314. [https://doi.org/10.1016/S0009-2541\(99\)00092-3](https://doi.org/10.1016/S0009-2541(99)00092-3)
- Wickham, H., Averick, M., Bryan, J., Chang, W., McGowan, L. D., François, R., et al. (2019). Welcome to the tidyverse. *Journal of Open Source Software*, 4(43), 1686. <https://doi.org/10.21105/joss.01686>
- Wickham, H., Pedersen, T. L., & Seidel, D. (2025). Scales: Scale functions for visualization.
- van Winden, J. F., Reichart, G.-J., McNamara, N. P., Benthien, A., & Damsté, J. S. S. (2012). Temperature-induced increase in methane release from peat bogs: A mesocosm experiment. *PLoS One*, 7(6), e39614. <https://doi.org/10.1371/journal.pone.0039614>
- Yarnes, C. (2013). $\delta^{13}\text{C}$ and $\delta^2\text{H}$ measurement of methane from ecological and geological sources by gas chromatography/combustion/pyrolysis isotope-ratio mass spectrometry. *Rapid Communications in Mass Spectrometry*, 27(9), 1036–1044. <https://doi.org/10.1002/rcm.6549>
- York, D. (1968). Least squares fitting of a straight line with correlated errors. *Earth and Planetary Science Letters*, 5, 320–324. [https://doi.org/10.1016/S0012-821X\(68\)80059-7](https://doi.org/10.1016/S0012-821X(68)80059-7)
- Yuan, K., Li, F., McNicol, G., Chen, M., Hoyt, A., Knox, S., et al. (2024). Boreal–Arctic wetland methane emissions modulated by warming and vegetation activity. *Nature Climate Change*, 14(3), 282–288. <https://doi.org/10.1038/s41558-024-01933-3>
- Yvon-Durocher, G., Allen, A. P., Bastviken, D., Conrad, R., Gudas, C., St-Pierre, A., et al. (2014). Methane fluxes show consistent temperature dependence across microbial to ecosystem scales. *Nature*, 507(7493), 488–491. <https://doi.org/10.1038/nature13164>
- Zalman, C. A., Meade, N., Chanton, J., Kostka, J. E., Bridgham, S. D., & Keller, J. K. (2018). Methylophilic methanogenesis in Sphagnum-dominated peatland soils. *Soil Biology and Biochemistry*, 118, 156–160. <https://doi.org/10.1016/j.soilbio.2017.11.025>
- Zhang, Y., Huang, X., Zhang, Z., Blewett, J., & Naafs, B. D. A. (2022). Spatiotemporal dynamics of dissolved organic carbon in a subtropical wetland and their implications for methane emissions. *Geoderma*, 419, 115876. <https://doi.org/10.1016/j.geoderma.2022.115876>
- Zhang, Z., Zimmermann, N. E., Stenke, A., Li, X., Hodson, E. L., Zhu, G., et al. (2017). Emerging role of wetland methane emissions in driving 21st century climate change. *Proceedings of the National Academy of Sciences*, 114(36), 9647–9652. <https://doi.org/10.1073/pnas.1618765114>