Science of the Total Environment 896 (2023) 165083

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Soil greenhouse gas emissions from drained and rewetted agricultural bare peat mesocosms are linked to geochemistry

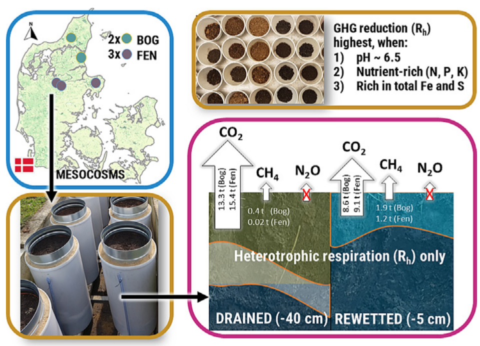
C.K. Nielsen [a,b,*], L. Elsgaard [a], U. Jørgensen [a,b], P.E. Lærke [a,b]

[a] Department of Agroecology, Faculty of Technology, Aarhus University, Blichers Alle 20, 8830 Tjele, Denmark
[b] CBIO, Centre for Circular Bioeconomy, Aarhus University, Denmark

HIGHLIGHTS

- For drained and rewetted peat, emissions were dominated by CO_2.
- GHG reduction highest on sites with: 1) pH 6.5, 2) rich in nutrients, iron, sulphur
- Rewetting of fen peatlands was most efficient.
- Biogeochemical proxies may be used for prioritising peatland areas for rewetting.

GRAPHICAL ABSTRACT



ARTICLE INFO

Editor: Jan Vymazal

Keywords:
Rewetting
Peatland
Greenhouse gases
Soil chemistry
Soil respiration

ABSTRACT

In view of climate considerations regarding the management of peatlands, there is a need to assess whether rewetting can mitigate greenhouse gas (GHG) emissions, and notably how site-specific soil-geochemistry will influence differences in emission magnitudes. However, there are inconsistent results regarding the correlation of soil properties with heterotrophic respiration (R_h) of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) from bare peat. In this study, we determined 1) soil-, and site-specific geochemical components as drivers for emissions from R_h on five Danish fens and bogs, and 2) emission magnitudes under drained and rewetted conditions. For this, a mesocosm experiment was performed under equal exposure to climatic conditions and water table depths controlled to either -40 cm, or -5 cm. For the drained soils, we found that annual cumulative emissions, accounting for all three gases, were dominated by CO_2, contributing with, on average, 99 % to a varying global warming potential (GWP) of 12.2–16.9 t CO_2eq ha^{-1} yr^{-1}. Rewetting lowered annual cumulative emissions from R_h by 3.2–5.1 t CO_2eq ha^{-1} yr^{-1} for fens and bogs, respectively, despite a high variability of site-specific CH_4 emissions, contributing with 0.3–3.4 t CO_2 ha^{-1} yr^{-1} to the GWP. Overall, analyses using generalized additive models (GAM) showed that emission magnitudes were well explained by geochemical variables. Under drained conditions, significant soil-specific predictor variables for CO_2 flux magnitudes were pH, phosphorus (P), and the soil substrate's relative water holding capacity (WHC). When rewetted, CO_2 and CH_4 emissions from R_h were affected by pH, WHC, as well as contents of P, total carbon and nitrogen. In conclusion, our results found the highest GHG reduction on fen peatlands, further highlighting that peat nutrient status and acidity, and the potential availability of alternative electron acceptors, might be used as proxies for prioritising peatland areas for GHG mitigation efforts by rewetting.

1. Introduction

The Paris Agreement was adopted by 196 Parties, among them the EU and its member states, at the COP 21 meeting in Paris (2015), with the

* Corresponding author at: Department of Agroecology, Faculty of Technology, Aarhus University, Blichers Alle 20, 8830 Tjele, Denmark.
E-mail address: claudia@agro.au.dk (C.K. Nielsen).

http://dx.doi.org/10.1016/j.scitotenv.2023.165083
Received 23 May 2022; Received in revised form 19 June 2023; Accepted 21 June 2023
Available online 29 June 2023

goal to reduce greenhouse gas (GHG) emissions by 40 % in 2030 as compared to the 1990 baseline and to reach zero-emissions by 2050 (Liobikienė and Butkus, 2017).

One critical tool in achieving this goal is the reduction of GHG emissions from drained peatlands. Peatlands are the dominating wetland type globally (Yu et al., 2010), covering about 3 % of the continental surface (Joosten and Clarke, 2002). A main feature of peatlands in their natural state is that the net primary production exceeds the organic matter decomposition (Lloyd et al., 2013), consequentially leading to the formation of peat, typically at a rate of ~1 mm yr^{-1} (Parish et al., 2008). This serves as a critical sink for atmospheric carbon dioxide (CO_2), but also provides the basis for other ecosystem services, such as nutrient retention, water filtration and habitat provision (Mitsch and Gosselink, 2000), thus highlighting the importance of natural peatlands at global and local scales.

However, anthropogenic degradation of peatlands by drainage for agriculture during the last century has led not only to a substantial loss of global peatlands (Joosten and Clarke, 2002) and the associated ecosystem functions (Bonn et al., 2016), but also to emission hotspots. Thus, peatlands release around 5 % of the total anthropogenic emissions in CO_2 equivalents (CO_2eq) from only 0.3 % of the total global land area (Joosten, 2016). Drainage, fertilisation and liming practices on peatlands cause the stimulation of microbial oxidative processes thereby enhancing heterotrophic respiration (R_h) (Limpens et al., 2008). Carbon (C) losses of up to 31 t CO_2-C ha^{-1} y^{-1} and nitrous oxide (N_2O) emissions of up to 35 kg nitrogen (N) ha^{-1} y^{-1} have been reported from northern agricultural peatlands (Maljanen et al., 2010). In Denmark, peatlands with an organic carbon content of >12 % cover 129,000 ha corresponding to approximately 3 % of the total land area (Greve et al., 2021). Of these, 57 % are drained and used for agricultural production, annually losing up to 16.7 t C ha^{-1} and up to 61 kg N_2O-N ha^{-1} y^{-1} (Elsgaard et al., 2012; Petersen et al., 2012) as a result of peat mineralisation.

Many studies have shown that rewetting of drained organic soils reduces emissions of CO_2 and N_2O (e.g., Komulainen et al., 1999; Strack and Zuback, 2013), while at the same time facilitating methane (CH_4) emissions due to limited soil aeration (Wilson et al., 2009). Data syntheses show that the net effect of rewetting results in lowered cumulative GHG emissions (Wilson et al., 2016). However, the site-specific climate mitigation potential by rewetting varies and nutrient-rich peatlands, such as the Danish riparian fens, might remain sources of cumulative emissions (in CO_2eq) even under wet conditions (Höper et al., 2008). In this context, little is known about the geochemical drivers and magnitudes of GHG emissions on the various types of drained agricultural peatlands in Denmark. While temperature (Elsgaard et al., 2012) and water table (Karki et al., 2014) are well known drivers for ecosystem respiration in temperate peatlands, soil-related drivers are rarely assessed. This is in particular true for the component of R_h, accounting for up to 50 % of total ecosystem respiration in wetlands (Wigand et al., 2009; Jovani-Sancho et al., 2018) and up to 90 % in forest soils (Hu et al., 2016). However, while assessments of whole-ecosystem GHG dynamics (i.e., including soil and plant respiration) are critical in order to account for total C budgets, are quantifications of R_h necessary to eliminating vegetation-related variations in GHG magnitudes, potentially adding bias to sound comparisons of soil-specific GHG mitigation potentials. Wilson et al. (2016) proposed a site-specific approach for assessing potential feasibilities of peatland rewetting, including site-specific physical and chemical soil properties. For the majority of studies assessing GHG dynamics following rewetting, standard assessments of peat properties include bulk density, total C and N, as well as pH (Loisel et al., 2014). However, there are inconsistent results regarding the link between soil-geochemical parameters and the emissions of CO_2, CH_4 and N_2O resulting from R_h from bare and cultivated peatlands. For instance, Weslien et al. (2009) reported high emissions of CH_4 from peat soils with a low pH in Sweden, whereas Ye et al. (2012) found enhanced CO_2 and CH_4 emissions from bogs and fens with a high pH level in Michigan, USA, due to enhanced fermentative activity in the peat. A study by Emsens et al. (2016) on Belgian and Dutch fen peatlands highlighted the negative influence of high soil iron (Fe) content on GHG mitigation by rewetting, which is due to enhanced Fe(III) mediated soil organic matter decomposition under oxygen limitation (Chen et al., 2020). Contrary to this, Saidy et al. (2020) reported lower emissions of CO_2 under the influence of high iron contents in peat.

In relation to the future management of Danish peatlands, there is a desiteratum to assess whether rewetting can mitigate high GHG emissions from agriculturally used, and notably how site-specific soil geochemistry will influence GHG magnitudes. Such an assessment might aid in pinpointing priority areas for rewetting to achieve the combined benefits of GHG mitigation, nutrient retention, and habitat creation.

We hypothesised that soil chemical and physical properties are drivers for differences in site-specific magnitudes of GHG emissions from drained peat soils and for the resulting mitigation potential by rewetting. The aims of this study were to 1) determine soil- and site-specific geochemical drivers for GHG emissions from R_h and their reduction potential by rewetting, 2) to determine magnitudes of emissions from soil respiration under controlled drained conditions, and 3) to assess the GHG mitigation potential from soil respiration by rewetting of different Danish agriculturally used peatlands.

2. Methods

2.1. Peatland sites and soil sampling

Intact soil samples were collected from two bog (B) peatlands: Store Vildmose (B_{SV}) and Lille Vildmose (B_{LV}), and three fen (F) peatlands: Selkær Enge (F_{SE}), Øby ($F_{ØB}$), and Vejrumbro (F_{VE}) (Fig. A.1). These sites represent part of the variation that nowadays is found in Danish bog and fen peatlands – both with regard to their original natural state and composition, and their prevailing peat depth and land-use history related to drainage and management (Table 1). Sampling occurred on areas that formerly were used for agricultural production, but now were classified as permanent grassland. All sites were shallow drained at the time of mesocosm collection.

In March 2019, ten intact soil mesocosms were collected from each of the five sites. Soil sampling was performed using PVC pipes (inner diameter, 30 cm; length, 60 cm) with sharpened edge that were mechanically

Table 1
Danish agricultural peatland donor sites, their peatland type and information regarding peat depth, land-use, drainage, and peat degradation (on the von Post scale, averaged over a soil profile of 100 cm depth) on their ecosystem- and field-scales.

Name	Type	Scale	Peat depth	Current land-use	Drainage	Von Post	Location	Area
Lille Vildmose (B_{LV})	Oligotrophic raised bog	Ecosystem:	1.5–5 m	Perennial grass, peat extraction	1760s - 2016		Northern Jutland	7800 ha
		Field:	1.5–2 m	Perennial grass (grazing)	1940s - 2016	H3	56°54′55″N, 10°12′28″E	6 ha
Øby ($F_Ø$)	Minerotrophic riparian fen	Ecosystem:	0–4 m	Perennial grass	1930s - 2018		Mid Jutland	139 ha
		Field:	~1.5 m	Perennial grass	1967–2018	H5	56°27′28“N, 9°40′32″E	6 ha
Selkær Enge (F_{SE})	Highly degraded minerotrophic fen (former bog)	Ecosystem:	0.2–1 m	Perennial grass	1850s – 2019		Mid Jutland	47 ha
		Field:	0.5–1 m	Perennial grass	1930s - 2019	H8	56°27′47“N, 10°44′38″E	6 ha
Store Vildmose (B_{SV})	Highly degraded oligotrophic raised bog	Ecosystem:	0.8–5 m	Perennial grass, potatoes, peat extraction	1920s - now		Northern Jutland	1900 ha
		Field:	1.5 m	Perennial grass (grazing)	1920s - now	H4	57°11′25″N, 9°49′4″E	5 ha
Vejrumbro (F_V)	Minerotrophic riparian fen	Ecosystem:	1–3 m	Perennial grass	1930s - now		Mid Jutland	106 ha
		Field:	~2 m	Perennial grass	1950s - 2022	H5	56°26′11“N, 9°33′00″E	10 ha

pressed into the soil and excavated. This resulted in a total of 50 mesocosms that were transported to open-air semi-field facilities at Aarhus University Viborg, Denmark.

2.2. Experimental design and set-up

The semi-field study was conducted from 21st of June 2019 to 20th of June 2020. Weather conditions during the study period are shown in Fig. A.2. Briefly, the monthly average temperature ranged between 4.0 and 16.8 °C, with August 2019 as the warmest month. Monthly average precipitation ranged between 32 and 134 mm, with April 2020 as the driest and February 2020 as the wettest month. The average soil temperature in 10 cm depth ranged between 3.5 °C in December 2019 and 17.7 in July 2019 (Fig. A.2).

The GHG mitigation potentials by rewetting of the different peat soils were studied with a generalized complete block design, with treatment (drained or rewetted) as block and site as a nested component. For this, the 10 intact soil mesocosms per site were trimmed by removal of vegetation and the by roots perfoliated upper topsoil layer (5 cm) before the mesocosms were set up in containers for controlled water table depth (WTD) adjustment in the semi-field facility. Removal of the vegetation was done to allow for dark chamber measurement of R_h in response to rewetting. A steel collar of 10 cm height was permanently placed on the rim of each mesocosm to allow tight placement of opaque chambers for GHG measurements without soil disturbance. The bare soil mesocosms were randomly grouped into two treatments: (i) drained (DRY) with a WTD of −40 cm and (ii) wet (WET) with a WTD of −5 cm. Each treatment had five replicates from each peatland site, resulting in a total of 25 mesocosms for each treatment. The mesocosms were placed in an outer plastic container, filled with gravel in the bottom on which the mesocosm was placed. WTD regulation was established by fitting overflow-tubes to the bottom site of the outer containers with the outlet-end cut at heights corresponding to the WTD treatment. The entire mesocosm setup, modified from Karki et al. (2015) and as schematically shown in Fig. A.3, was placed in two separate ditches (1 m deep x 1 m wide x 8 m long) for each WTD treatment. The soil surface was at ground level elevation and the mesocosms were insulated with seashells to secure robust soil temperature conditions. The mesocosms were rewetted to maintain the selected WTDs using demineralised water, supplied by drip irrigation into the outer containers in intervals of 1 h once every third day during winter months (October to March) to 1 h twice daily during summer months (April to September). Five soil mesocosms per treatment, one for each peatland site, were equipped with a piezometer, time domain reflectometry (TDR) probes, and a soil temperature probe at −10 cm depth (these mesocosms were not used for GHG measurements). The relative water holding capacity (WHC) was measured during each GHG sampling campaign for each site and treatment with the TDR probes. Maximum fluctuations of WTD, due to events of heavy rain or high evapotranspiration, were ± 1.3 cm (DRY) and ± 2 cm (WET), resulting in annual averaged WTDs of - 40.4 cm and - 5.0 cm, respectively. The mesocosms received no fertilisation and were permanently kept free from emerging and invading vegetation by manual weeding.

2.3. Soil physical and chemical properties

From each of the studied peatlands, three soil core samples of 5 cm in diameter and 60 cm depth were taken using a Russian peat corer for analysis of soil chemical and physical properties. Soil analyses were performed for the following soil segments: 0–5 cm, 5–20 cm, 20–40 cm, and 40–60 cm. Soil bulk density (BD) was determined by oven drying at 60 °C to constant weight, and soil pH was measured after soaking 5 g of dry soil in 50 mL H_2O overnight. Total N and C were analysed on a Vario MAX CN (Elementar Analysesysteme GmbH, Hanau, Germany). The contents of Fe, potassium (K), phosphorus (P) and sulphur (S) were determined by inductively coupled plasma optical emission spectroscopy (ICP-OES) as described by Liu et al. (2013). A van Post classification of peat humification (von Post, 1922; von Post and Granlund, 1926) was performed for soil segments of 10 cm in the upper 20 cm and segments of 20 cm in the remaining 60 cm.

2.4. GHG measurements and flux calculation

Measurements of CO_2, CH_4 and N_2O fluxes from R_h took place every two weeks in the study period, resulting in 27 sampling campaigns, performed between 9:30 and 14:30, using opaque PVC chambers with a volume of 41.89 L (diameter, 30 cm; height, 50 cm). For each campaign, the sampling order was randomised in order to account for diurnal variability. The chambers were equipped with a fan to mix headspace air, a vent to ensure pressure equilibrium, and a temperature probe measuring air temperature inside the chamber. Five gas samples per chamber were withdrawn at 0, 5, 10, 25, and 50 min after chamber closure using a syringe (20 mL), connected to the chamber sampling port by a polypropylene tube of 1.2 m length and 4 mm inner diameter. To remove dead volume of the tubing, 16 mL air from the tube and air sampling system were removed with a syringe and discarded prior to taking the gas samples. Eventually, 11 mL of air from the headspace was transferred to pre-evacuated 6 mL glass exetainers (Labco Limited, UK). The exetainers were stored for less than four weeks until gas analysis on an Agilent 7890 gas chromatograph (GC), equipped with an automatic injection system (CTC CombiPAL, Agilent A/S, Nærum, Denmark). Details of the GC and instrument calibration were described by Petersen et al. (2012).

Gas fluxes were calculated using linear regression within the package *flux* (Jurasinski et al. (2014) version 0.3–0) in the statistical program R (R Core Team (2020) Version 4.1.2 – "Bird Hippie"), based on best linear fits according to the smallest normalised root mean square error (NRMSE), calculated on at least four out of the five gas concentration measurements. Additional quality checks for the individual fluxes included R^2, a defined quality range for concentration measurements, the number of measurements below ambient (NOMBA) as well as a GC quality flag to indicate observed errors resulting from the GC analysis. Further, fluxes of CH_4 and N_2O were tested for chamber leakage or gas sampling errors based on a control against CO_2 concentration data, resulting in no fluxes discarded. CO_2 fluxes were accepted when all three assumptions were met: 1) no quality flags were violated, 2) $R^2 > 0.95$, and 3) significant regression for flux estimation ($p < 0.05$). Due to frequent CH_4 and N_2O fluxes near zero, fluxes were accepted when the following criteria were met: 1) no quality flags were violated, and 2) $R^2 > 0.90$. Further, fluxes of all gases beyond the 2.0 percentile tail of a log-norm-distribution were regarded as outliers and discarded ($n = 29$). In addition, nine CH_4 fluxes indicating an uptake rate >0.5 mg m^{-2} h^{-1} were discarded according to Hütsch (2001) who found no CH_4 uptake rates >0.3 mg m^{-2} h^{-1} in any ecosystems. Hence, out of initially 1080 fluxes for each gas, 90 %, 79 % and 90 % of measured CO_2, CH_4, and N_2O fluxes, respectively, met our quality criteria and were included in all further calculations and analyses. Summer fluxes were defined as those in the period from April to September, whereas winter fluxes were those from October to March.

2.5. Interpolation to annual gas fluxes

Fluxes of CO_2 from all mesocosms and treatments were interpolated to annual values using the function *budget.reco* in the *flux* package (Jurasinski et al., 2014). Based on the relationship between soil temperature and measured fluxes, hourly values for CO_2 were modelled using an exponential regression model of Arrhenius-type, as modified by Lloyd and Taylor (1994), with hourly soil temperature from the semi-field facility as driver (Eq. (1)):

$$R = \theta_1 * \exp\left(E_0 * \left(\frac{1}{T_{ref} - T_0} - \frac{1}{Temp - T_0}\right)\right) \quad (1)$$

where R is the soil respiration flux rate, θ_1 is the base respiration rate (in μmol C m^{-2} s^{-1}) at the reference temperature (T_{ref}) of 10 °C, E_0 is the parameter of temperature sensitivity of *R*, T_0 is the theoretical lower temperature limit for the soil respiration (here constrained to -46.02 °C) and *Temp* is the soil temperature (°C). Hourly values were summed up to determine annual CO_2 emissions as well as summertime and wintertime emissions.

Annual cumulative balances of CH_4 were calculated by modelling hourly values using a simple linear regression model, based on soil temperature fits as applied for CO_2:

$$R = \theta_1 * \theta_2 * T \tag{2}$$

where *R* is the soil respiration flux rate, θ_1 is the base respiration rate (in μmol C m^{-2} s^{-1}) at the reference temperature (T_{ref}) of 10 °C, θ_2 is the bias coefficient of 0.08 ± 0.01 and *Temp* is the soil temperature (°C).

Both CO_2 and CH_4 interpolation models were quality checked regarding their fit by posteriori analyses of model performance and passed at $p < 0.01$ and $R^2 > 0.8$.

Annual N_2O emissions for each mesocosm and treatment were calculated following a Monte Carlo permutation procedure, as described in detail by Huth et al. (2013). Annual CH_4 and N_2O fluxes were converted into CO_2eq using the 100-yr global warming potentials (GWP) of 28 and 265, respectively (Myhre et al., 2013; Evans et al., 2021).

2.6. Environmental variables, soil moisture and water table depth

Hourly average air temperatures during GHG sampling campaigns, daily temperature means, and daily average precipitation were gathered from a meteorological station located 1.3 km from the semi-field site. Soil temperature within the mesocosms was automatically logged hourly during the entire study period. During each GHG sampling campaign, water table depth (WTD) was monitored in the pre-installed piezometers (one replicate per site and treatment), while the electrical conductivity to obtain WHC was measured in the soil mesocosms with the pre-installed TDR probes.

2.7. Statistical analysis

To estimate the influence of covariates, including biogeochemical properties, on gas fluxes of CO_2 and CH_4, we used generalized additive models (GAMs) with a Gaussian distribution and an identity link function for the log or square root transformed response variables, thereby fitting mean and variance on the log scale. The usage of GAMs was chosen, since GAMs allow for an estimation of non-linear and smooth relationships in a flexible modelling approach based on individual data-derived penalties (Marra and Wood, 2011; Wood, 2011; Wood et al., 2016). GAMs were performed with the package *mgcv* (Wood, Version 1.8–38, 2021) in R, using restricted marginal likelihood for all model coefficients and penalties, in which the following model was used:

$$y_i \sim N\left(\mu, \sigma^2\right)$$

$$\begin{aligned} y_i = {} & \alpha + f_1(t.air_i, t.soil_i) + f_2 intrannual(date_i) + f_3(tdr_i) + f_4(S_i) \\ & + f_5(Fe_i) + f_6(P_i) + f_7(pH_i) + f_8(BD_i) + f_9(TC_i) + f_{10}(TN_i) \\ & + f_{11}(K_i) + \beta_1 x_1 + \varepsilon_i, \varepsilon_i \sim N\left(\mu, \sigma^2\right). \end{aligned}$$

where y_i is the log or square root transformed observed dependent variable and μ is the overall mean, affected by f_1: the isotropic product smooth representing the marginal effects and interaction of temperature (t.air and t. soil), f_2 : intrannual(date), i.e., the penalised cyclical smooth term of data where spline end points are equal, $f_3 - f_{11}$: smooth functions of the various covariates for the *i*th sample, and x_1, the categorical predictor variable of peatland type (bog or fen). α is the intercept and σ^2denotes the experimental error. Prior to model fitting, variables were controlled for multicollinearity by estimating the Pearson product moment correlation coefficient. The final model was checked for concurvity and model residuals were inspected for normality and homoscedasticity. To test for significance of differences between means, a one-way ANOVA with post-hoc Tukey HSD at 95 % confidence level was performed. Correlation effects between site specific soil properties and the various annual cumulative GHG balances and reduction potentials by rewetting were determined by multiple linear regression using Pearson's correlation. Measures of central tendency are given as means and dispersion around the means are given as standard error ($n = 4$) unless otherwise specified. For all statistical analyses, averaged soil properties for the depth of -5 cm to -25 cm were used.

3. Results

3.1. Soil properties

Content gradients (in g per kg) with depth were observed for all soil chemical and physical parameters (Table A.1), which in particular were pronounced for TC, K, and P. TC was lowest in fen soils and highest in bog soils and ranged on average from 35.2 % (F_V) to 51.3 % (B_{SV}) across depth. In contrast, TN was highest in fen peat soils (2.6–2.9 %) and lowest in bog peat soils (1.3–1.4 %). TC increased with depth for all sites, whereas TN showed a decline for bogs and was stable for fens. Across depth in the fen peat soils, the average contents of Fe, P and S (kg^{-1} DM) ranged from 8.0 to 10.7 g Fe, 1.1–1.3 g P, and 7.1–11.5 g S, whereas the bog peat soils showed 2–10 times lower contents. Averaged pH ranged from 4.6 to 6.4 and differed significantly ($p < 0.001$) between fens and bogs, with bogs being more acidic than fens.

3.2. Fluxes of GHG from heterotrophic respiration

3.2.1. Measured and modelled CO_2 fluxes

Mid-day heterotrophic soil respiration across all sites was 52 % lower in WET as compared to DRY on an average annual basis (Fig. 1A). Flux magnitudes from fen peatlands were highly similar. Soil from F_{SE} showed the highest emission among the five DRY sites (237 ± 37 mg CO_2 m^{-2} h^{-1}), whereas F_V had the highest emissions among WET treatments (128 ± 26 mg CO_2 m^{-2} h^{-1}). For treatment DRY, the lowest average fluxes over the sampling period (133 ± 26 mg CO_2 m^{-2} h^{-1}) were detected from B_{SV}. For treatment WET the lowest average fluxes were from B_{SV} (101 ± 28 mg CO_2 m^{-2} h^{-1}). Under drained conditions, fen peatland soils emitted on average 214 ± 38 mg CO_2 m^{-2} h^{-1}, which was 37 % more ($p < 0.001$) than bog peatlands (156 ± 30 mg CO_2 m^{-2} h^{-1}). For treatment WET, there was no difference in CO_2 emissions between the five soil types, with the exception of F_V, showing on average 34 % higher soil respiration as compared to the other sites. Summer fluxes (mean temperature: 13.1 °C) across all sites were on average 54 % (WET) - 65 % (DRY) higher as compared to winter fluxes (mean temperature: 5.3 °C), indicating the high temperature dependency. Treatment WET reduced average CO_2 fluxes during the summer months by 40 % (B_{SV}) – 72 % (F_{SE}) as compared to DRY. The reduction of annual average CO_2 flux by raising the WTD from -40 cm to -5 cm ranged between 38 % (B_{SV}) to 69 % (F_{SE}). Modelled hourly rates of (R_h) showed a strong seasonality (Fig. A.4), with low and near-zero fluxes during winter months for all soils on treatments DRY and WET.

3.2.2. CH_4 fluxes

Annual methane fluxes were generally low for all sites in treatment DRY, with B_{SV} showing the highest annual emissions (0.4 ± 0.3 mg CH_4 m^{-2} h^{-1}). Dry soils from B_{LV} and $F_Ø$ were characterised by a minor uptake of CH_4. Rewetting raised annual fluxes significantly ($p < 0.001$) for all soils (Fig. 1B) with, on average, similar resulting fluxes from fens (0.7 ± 0.6 mg CH_4 m^{-2} h^{-1}) and bogs (0.6 ± 0.4 mg CH_4 m^{-2} h^{-1}). However, we observed high variabilities across sites of the same peatland type. For instance, annual methane emissions for fens were highest from F_V (1.3 ± 1.1 mg CH_4 m^{-2} h^{-1}), as compared with $F_Ø$ and F_{SE} (means, <0.5 mg CH_4 m^{-2} h^{-1}). For bogs, B_{SV} emitted 1.1 ± 0.8 mg CH_4 m^{-2} h^{-1}, which was significantly

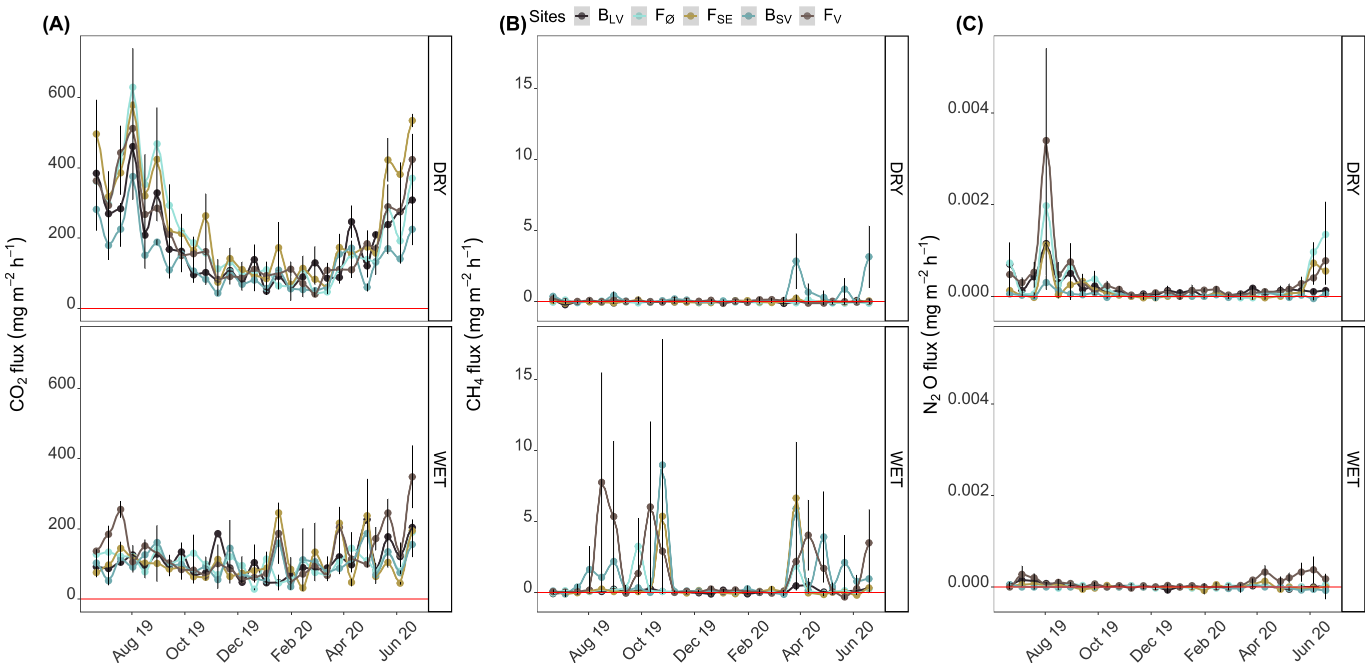


Fig. 1. Measured fluxes (in mg m^{-2} h^{-1}) for the study sites Lille Vildmose (B_{LV}), Øby ($F_Ø$), Selkær Enge (F_{SE}), Store Vildmose (B_{SV}) and Vejrumbro (F_V) of A) carbon dioxide (CO_2), B) methane (CH_4) and C) nitrous oxide (N_2O) for the dry (top panels) and wet (bottom panels) treatments. Grey shaded lines for the various soil types show averaged flux values at the specific sampling campaigns from the replicates ($n = 4$) with standard errors showing the variation. Red lines indicate zero.

($p < 0.001$) more than B_{LV} (0.1 ± 0.1 mg CH_4 m^{-2} h^{-1}). Summer emissions of CH_4 at WET were for B_{LV}, $F_Ø$ and F_V higher than those during the colder winter half-year. However, due to high methane emissions (>5.4 mg CH_4 m^{-2} h^{-1}) on the 25.10.2019 and 27.03.2020, the integrated rates were highest during winter for soils from F_{SE} and B_{SV}.

3.2.3. N_2O fluxes

Nitrous oxide fluxes were generally low and close to zero in the majority of occasions, with the highest overall flux of 3.4 ± 2.0 μg N_2O m^{-2} h^{-1} observed for soils from F_V on treatment DRY. N_2O fluxes for DRY showed an annual average between 0.04 ± 0.04 (B_{SV}) and 0.34 + 0.18 μg N_2O m^{-2} h^{-1} (F_V), with the highest contribution during summer months (Fig. 1C). Rewetting reduced annual average N_2O fluxes by 72 % (F_V) to 97 ($F_Ø$) % or even resulted in small net N_2O uptake (B_{SV}), resulting in fluxes between −0.01 ± 0.03 (B_{SV}) and 0.10 ± 0.06 (F_V) μg N_2O m^{-2} h^{-1}.

3.3. Annual balances and the GWP of soil respiration

Annual cumulative emissions from R_h, when accounting for all three gases, were for all DRY soils significantly dominated by CO_2, ranging between 11.4 ± 0.8 (B_{SV}) and 16.8 ± 0.6 (F_{SE}) t CO_2 ha^{-1} yr^{-1} (Table 2). The contribution of N_2O to the GWP was below 0.01 % and hence negligible. CH_4 contributed with 0.8 ± 0.5 t CO_2eq ha^{-1} yr^{-1} most to the cumulative annual emissions from B_{SV} while soils from B_{LV} and $F_Ø$ were characterised by a minor uptake of CH_4. On average, drained bog sites emitted 13.7 % less CO_2eq ha^{-1} yr^{-1} than fens (Table 3). Rewetting raised the share of CH_4 to total emissions on all sites, thus contributing 3 % (B_{LV}) – 30 % (B_{SV}) of the overall GWP, while CO_2 emissions were significantly ($p < 0.001$) lowered by 3.3 (B_{SV}) to 9.2 (F_{SE}) t CO_2 ha^{-1} yr^{-1}. The contribution of N_2O to the GWP was even lower under rewetted than drained conditions. When accounting for all three gases, annual cumulative emissions in t CO_2eq ha^{-1} on treatment DRY ranged from 12.2 ± 1.2 (B_{SV}) to 16.9 ± 0.6 (F_{SE}). Rewetting reduced the annual cumulative emissions of all soils by 6 % (B_{SV}) to 48 % (F_{SE}) to values ranging from 8.7 ± 1.1 (F_{SE}) to 12.6 ± 1.4 (F_V) t CO_2eq ha^{-1} yr^{-1}. However, there was no statistical difference in annual cumulative emissions between fen and bog peatlands on both treatments DRY and WET (Fig. 2). Nonetheless, significant reductions of cumulative emissions from R_h following rewetting were found for soils from B_{LV}, $F_Ø$ and F_{SE}, and also across fen peatlands (Table 4). Generally, we

Table 2
Average annual cumulative emissions of carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O) as well as their summed global warming potential for a 100-year time horizon (GWP). All values in t CO_2eq ha^{-1} yr^{-1} with a global warming potential conversion factor of 28 for CH_4 and 265 for N_2O. Values are given for soil mesocosms from Lille Vildmose (B_{LV}), Øby ($F_Ø$), Selkær Enge (F_{SE}), Store Vildmose (B_{SV}) and Vejrumbro (F_V) for each of the treatments: dry and bare (DRY) and wet and bare (WET). Standard error is reported in brackets. Letters indicate differences between means, where treatments with the same letter are not significantly different.

	DRY					WET				
	B_{LV}	$F_Ø$	F_{SE}	B_{SV}	F_V	B_{LV}	$F_Ø$	F_{SE}	B_{SV}	F_V
CO_2	15.13 (± 1.37)ab	14.07 (± 1.17)ab	16.82 (± 0.64)a	11.40 (± 0.75)b	15.22 (± 1.56)ab	9.16 (± 0.2)a	8.90 (± 0.45)a	7.65 (± 0.92)a	8.09 (± 1.09)a	10.77 (± 0.86)a
CH_4	−0.01 (± 0.03)a	−0.04 (± 0.03)a	0.03 (± 0.04)a	0.75 (± 0.47)a	0.08 (± 0.07)a	0.32 (± 0.22)a	0.64 (± 0.32)a	1.09 (± 0.73)a	3.40 (± 2.06)a	1.82 (± 0.52)a
N_2O	0.004 (± 0.001)a	0.005 (± 0.002)a	0.003 (± 0.001)a	0.001 (± 0.001)a	0.008 (± 0.003)a	<0.001 (± <0.001)ab	<0.001 (± <0.001)b	<0.001 (± <0.001)b	<0.001 (± <0.001)b	0.002 (± 0.001)a
GWP	15.13 (± 1.36)a	14.03 (± 1.15)a	16.85 (± 0.62)a	12.16 (± 1.15)a	15.31 (± 1.52)a	9.48 (± 0.31)a	9.54 (± 0.7)a	8.73 (± 1.14)a	11.49 (± 2.18)a	12.59 (± 1.37)a

Table 3

Average annual cumulative emissions of carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O) as well as their summed global warming potential for a 100-year time horizon (GWP). All values in t CO_2eq ha^{-1} yr^{-1} with a global warming potential conversion factor of 28 for CH_4 and 265 for N_2O. Values are given as means across sites per peatland type for each of the treatments: dry and bare (DRY) and wet and bare (WET). Standard error is reported in brackets. Letters indicate differences between means, where treatments with the same letter are not significantly different. The average reduction of cumulative emissions per gas from WET as compared to DRY is given as a negative (−) or positive (+) change in percentage, without variation.

	DRY		WET		Change (%)	
	Bog	Fen	Bog	Fen	Bog	Fen
CO_2	13.27 (± 1.01) a	15.37 (± 0.71) a	8.62 (± 0.55) a	9.11 (± 0.56) a	- 35.04	- 40.73
CH_4	0.37 (± 0.26) a	0.02 (± 0.03) a	1.86 (± 1.12) a	1.18 (± 0.32) a	+ 402.70	+ 5800
N_2O	<0.001 (± < 0.001) a	0.001 (± < 0.001) a	<0.001 (± < 0.001) a	<0.001 (± < 0.001) a	0	0
GWP	13.64 (± 1.0) a	15.40 (± 0.7) a	10.49 (± 1.09) a	10.29 (± 0.76) a	- 23.09	- 33.18

observed high variations of GWP across replicates for the various sites and peatland types for both the DRY and WET treatments (Fig. 3A). The contribution of CO_2 to the within-site variation (Fig. 3B) was for both treatments more pronounced than the contribution from CH_4 (Fig. 3C).

3.4. The effect of site-specific soil properties and environmental variables on GHG fluxes

3.4.1. Drained conditions

For both CO_2 and CH_4 fluxes, the GAM analysis showed that fluxes from treatment DRY, were well explained by peatland type (p < 0.001) (Table A.2). Further, flux magnitudes were significantly (p < 0.001) affected by the WHC, soil and air temperatures, and date. Soil specific predictor variables of high importance for CO_2 were pH (p < 0.001), P (p < 0.01), and to a lesser extent TN (p < 0.05). The GAM was able to explain 92 % of the CO_2 variation, showing that the lowest CO_2 fluxes under drained conditions occurred from peatland sites where pH, P, TN and WHC were low (Fig. B.1). For CH_4, the GAM detected that site-specific BD (p < 0.001), K, TN and pH (all p < 0.01) were important predictor variables for flux magnitudes. However, contrary to fluxes of CO_2, the highest CH_4 fluxes occurred at sites with low pH but higher BD, high K, and high TN (Fig. B.2). Aside from pH, our findings also demonstrated such contrasting behaviour between drivers for CO_2 and CH_4 fluxes for the isotrophic smooth interaction of soil and air temperatures. Here the GAM indicated that a rise in air temperature affected magnitudes of CO_2 emissions, while fluxes of CH_4 showed a higher dependency on soil temperatures (Figs. B.3, 4). Overall, the GAM for CH_4 fluxes under drained conditions allowed us to detect non-linear trends with 83 % of the deviance explained.

3.4.2. Rewetted conditions

Concerning fluxes of CO_2 for treatment WET, the GAM explained 78 % of the deviance (Table A.3). As also for DRY, the smooth predictor variables of temperature, date, WHC, as well as the parametric coefficients of peatland type, had a significant influence (all p < 0.001) on flux magnitudes. In addition, CO_2 fluxes displayed a negative response with increasing pH (p < 0.001), while fluxes increased with higher P contents (p < 0.001) and lower WHC (p < 0.001) (Fig. B.5). Regarding CH_4, we were able to explain 75 % of the deviance, with bog peatland type, WHC, temperatures, date, and TN (all p < 0.001), as well as pH (p < 0.05) and TC (p < 0.01) affecting emission magnitudes. The GAM analysis showed that high CH_4 fluxes under rewetted conditions occurred predominantly on soils with high contents of TC (p < 0.001) but low TN (p < 0.001), and low pH (p < 0.05) (Fig. B.6). Also, for the WET treatments, the GAMs detected similar, though not as pronounced, contrasting responses of CO_2 and CH_4 fluxes to air and soil temperatures as under drained conditions (Figs. B.7, 8).

3.4.3. Correlation of soil properties and annual cumulative emissions

The Pearson's correlation detected significant correlations between average soil properties at the depth of −5 cm to −25 cm and average annual GHG emissions from R_h across the different peatland sites for treatments DRY and WET (Table A.4), with the majority of correlations between the various annual cumulative emissions and GWP. Annual CO_2 emissions

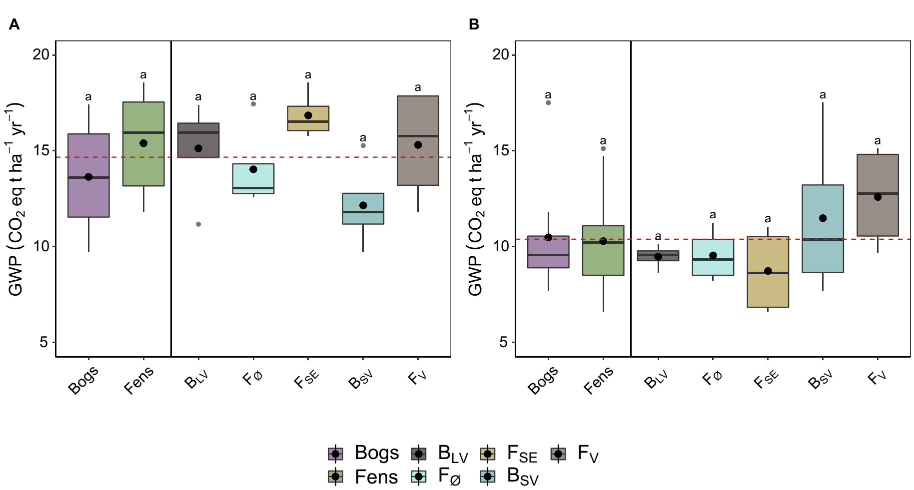


Fig. 2. Boxplot showing the global warming potential (GWP) in annual cumulative emissions (t CO_2eq ha^{-1} yr^{-1}) for the study sites Lille Vildmose (B_{LV}), Øby ($F_Ø$), Selkær Enge (F_{SE}), Store Vildmose (B_{SV}) and Vejrumbro (F_V), as well as across the peatland types bog and fen for the treatments DRY (panel A) and WET (panel B). Bold black dots represent the mean for each site, based on four replicates, while the interior line represents the median. The boxes bound the interquartile range (IQR) (25th to 75th quartile). The whiskers extend to 1.5 times the IQR, and their ends show the highest and lowest value excluding outliers, which are represented by dark grey dots. Red dashed lines indicate averages across all sites and letters indicate differences between means, where treatments with the same letter are not significantly different.

Table 4
Table showing significant differences of annual cumulative emissions of carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O) as well as their summed global warming potential for a 100-year time horizon (GWP) between the treatments dry and bare (DRY) and wet and bare (WET) for soil mesocosms from Lille Vildmose (B_{LV}), Øby ($F_Ø$), Selkær Enge (F_{SE}), Store Vildmose (B_{SV}) and Vejrumbro (F_V) as well as per peatland type. Letters indicate differences between means, where treatments with the same letter are not significantly different.

	Bog		Fen		B_{LV}		$F_Ø$		F_{SE}		B_{SV}		F_V	
	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet
CO_2	a	b	a	b	a	b	a	b	a	b	a	b	a	b
CH_4	a	a	b	a	a	a	a	a	a	a	a	a	b	a
N_2O	a	b	a	b	a	b	a	a	a	a	a	a	a	a
GWP	a	a	a	b	a	b	a	b	a	b	a	a	a	a

were for both treatments correlated with N_2O. The GWP was highly correlated with all other annual cumulative values for the various gases on treatment WET, while no correlations were found for DRY. For CH_4, the only significant ($p < 0.001$) relationship was found with the GWP on treatment WET.

Regarding correlations of GHG with average soil properties, we only found minor significant ($p < 0.05$) correlations for DRY between CO_2 and the soil properties BD, Fe, and TC. The negative correlation between CO_2 and TC was also observed between GWP (DRY) and TC.

4. Discussion

The present study assessed differences in soil respiration for different Danish agriculturally used peatlands under drained (WTD, −40 cm) and rewetted (WTD, −5 cm) conditions, focusing on potential site-specific geochemical drivers for emissions resulting from R_h and their effects on the GHG mitigation potential by rewetting. As expected, and widely known from previous research, CO_2 and CH_4 fluxes for both treatments were strongly affected by environmental factors, in particular ambient temperatures, as well as the soil-specific WHC. However, for both gases were not only those covariates, but, in

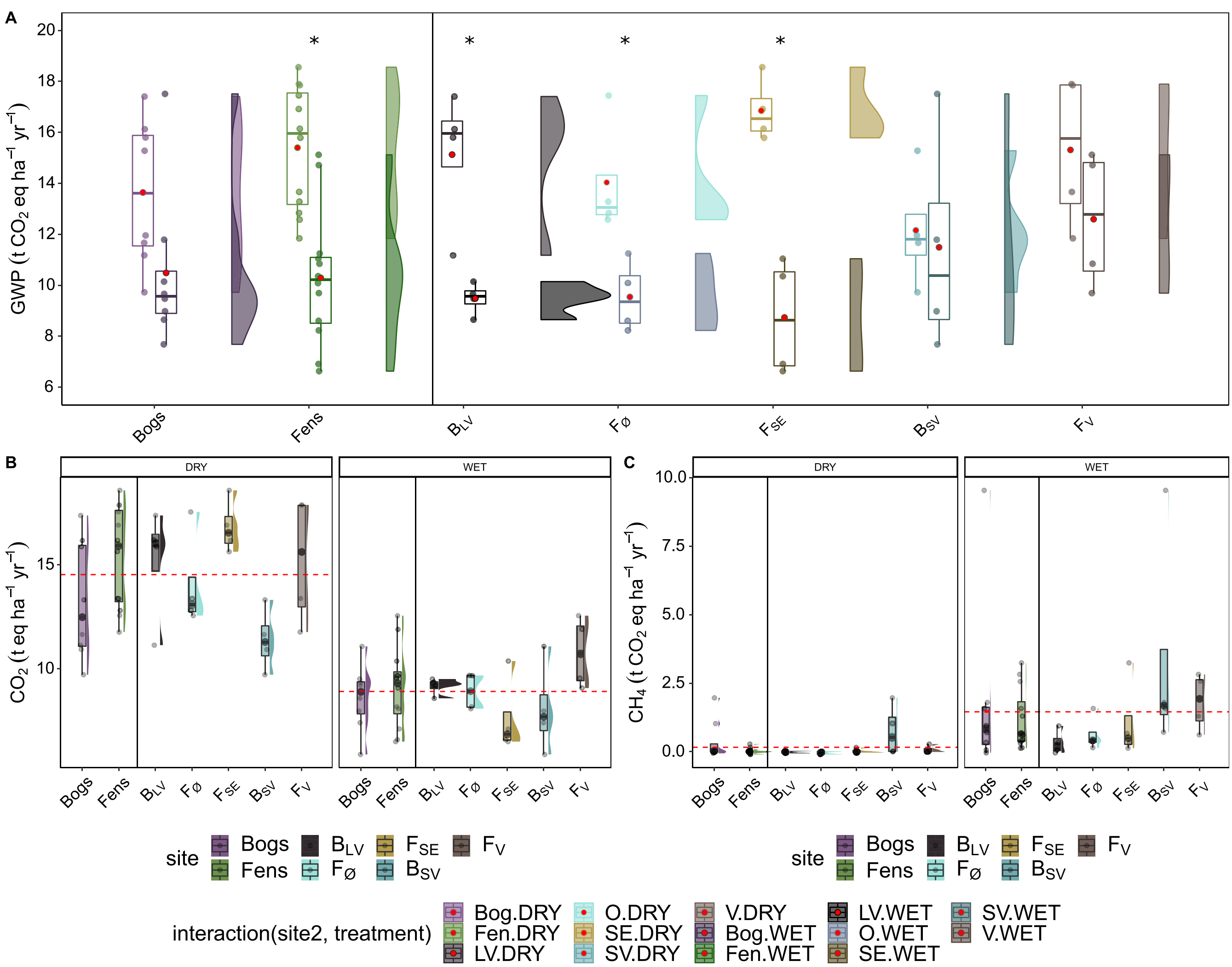


Fig. 3. Combined raincloud and boxplots showing the within and in-between site variation of annual cumulative greenhouse gas emissions, based on the individual contributions of CO_2 and CH_4 per replicate. Panel A): Vertical raincloud plot illustrating the data distribution (the "cloud") with jittered raw data (the dots) from each replicate, and data statistics (the boxplots) for the study sites Lille Vildmose (B_{LV}), Øby ($F_Ø$), Selkær Enge (F_{SE}), Store Vildmose (B_{SV}) and Vejrumbro (F_V), as well as across the peatland types bog and fen. Paler colours show the global warming potential (GWP) in annual cumulative emissions (t CO2eq ha^{-1} yr^{-1}) for treatment DRY, while darker colours show the GWP for treatment WET. Stars denote significant differences between treatments for the various sites. Panel B): Boxplot showing annual cumulative emissions of carbon dioxide (CO_2) in t CO_2 ha^{-1} yr^{-1} for the various sites and types for treatment DRY and WET. Panel C): Boxplot showing annual cumulative emissions of methane (CH_4) in t CO_2eq ha^{-1} yr^{-1} for the various sites and types for treatment DRY and WET. In all boxplots on the panels, bold red or black dots represent the mean for each site, based on four replicates, while the interior line represents the median. The boxes bound the interquartile range (IQR) (25th to 75th quartile). The whiskers extend to 1.5 times the IQR, and their ends show the highest and lowest value excluding outliers, which are represented by colored (panel A) or dark grey (panels B and C) dots. Red dashed lines indicate averages across all sites.

particular, site-specific soil properties detected as critical by the GAM models.

4.1. Emission reductions across sites were similar following rewetting

The spatial variability of GHG fluxes and annual cumulative emissions in drained and rewetted agricultural and natural peatlands is to a large extend controlled by environmental factors, such as temperature (Elsgaard et al., 2012), seasonal patterns of precipitation (Barel et al., 2021), vegetation communities (Couwenberg et al., 2011), and groundwater levels (Evans et al., 2021). However, most studies so far focused on assessing the controls on a site-specific GWP under drained and rewetted conditions, of either 1) net ecosystem exchange or net ecosystem carbon balance (e.g., Tiemeyer et al., 2020); 2) those of R_h but in differing climatic zones, such as the tropics (e.g., Hergoualc'h et al., 2017), or 3) with a focus on the effects of abiotic factors (e.g., Järveoja et al., 2020). Thus, a sound comparison of the site-specific values for R_h obtained in this study to those from other sites in similar climatic settings is difficult to perform, in particular regarding fen peatlands where emission factors from bare peat soil are scarce.

Nonetheless, one option is to examine reported emission factors for drained bog peatlands undergoing active peat extraction for e.g., horticulture. Both bogs assessed in our study have been subject to peat extraction for horticulture and fuel in the past, with activities ceasing last on B_{SV}. Hence, comparing the GWPs for these peatlands with the range of reported values for active peat extraction sites (4.0–16.0 t CO_2eq ha^{-1} yr^{-1} (IPCC, 2014); 11.3 t CO_2eq ha^{-1} yr^{-1} (Wilson et al., 2016); 2.6–5.0 t CO_2 ha^{-1} yr^{-1} (Shurpali et al., 2008)), showed that observed R_h was in the upper range, with the main contributor to the overall GWP being CO_2. However, both bogs also have shown differences regarding their GHG emission potentials under drained and rewetted conditions. While B_{LV} was characterised by a GWP of 15.1 ± 1.3 t CO_2eq ha^{-1} yr^{-1} under drained conditions, which was significantly ($p < 0.001$) reduced by 37 % following rewetting, were cumulative annual emissions from R_h for B_{SV} similar, independent of the soils water saturation. This discrepancy between the potential for emission reduction might be found in their site-specific record regarding duration and, therefore, depth of peat extraction, affecting the intrinsic recalcitrance of the peat substrate (Kleber, 2010) which also has been recently highlighted by Clark et al. (2023).

Nonetheless, our study under equal environmental conditions, and controlled WTDs for all soil sites, only showed significant differences of flux magnitudes and annual cumulative CO_2 emissions between F_{SE} and B_{SV}. Beyond that, emission magnitudes and budgets from R_h, were similar across sites, peatland types, and stages of water saturation. Further, although original vegetation, including top-soil roots, has been removed for the mesocosms in this study, decomposing fine roots in the deeper soil layers were likely to have caused the partially high CO_2 emissions in the drained state (Iversen et al., 2018; Dezzeo et al., 2021).

However, while the contribution of CO_2 from WET to the annual GWP was similar across sites, we observed considerable differences regarding CH_4 emissions from R_h, ranging from 0.3 (B_{LV}) to 3.4 (B_{SV}) t CO_2eq ha^{-1} yr^{-1}. In fact, from B_{LV} and $F_Ø$ they even were as low as CH_4 emissions reported following topsoil removal and sphagnum introduction (Huth et al., 2020), both associated with low CH_4 emissions (Larmola et al., 2010). In our study, the removal of the upper, by roots perfoliated, topsoil layer might contribute with a causation to the low observed CH_4 fluxes at high WTD. In addition, years of drainage or peat extraction might contribute to a potential lag-time effect regarding the potential for CH_4 recovery as seen on the example of B_{SV}, expressing the potential for methanogenic activity at the end of the study period and following heavy rainfall events. Nonetheless, differences and irregularities of CH_4 emissions for drained and rewetted peatlands have already been reported elsewhere (e.g., Vybornova et al., 2019), thus highlighting the importance of peat substrate quality for the determination of soil related GHG mitigation potentials.

Nonetheless, contrary to most other studies, we were able to measure GHG emissions from R_h at permanently controlled WTD levels, thereby being independent from environmental fluctuations, usually significantly affecting fluxes in natural ecosystems and agroecosystems (Evans et al., 2021). Further, due to our study set-up, the soil mesocosms were equally exposed to ambient temperatures, known to affect CH_4 production and oxidation potentials (Zhang et al., 2021). Being able to eliminate these critical factors for site-specific differences in emission magnitudes, the question regarding a causation for observed differences in CH_4 emissions, both between and within sites, remains, though the divergent nature of CH_4 fluxes in wetlands is commonly acknowledged (Knox et al., 2019). We observed this strong variability also across replicates for the various sites and peatland types for both, the DRY and WET treatments, confirming the transient and divergent nature of methane even on micro-spatial scale.

Across peat soil types, the average emission reduction of 4.1 t CO_2eq ha^{-1} yr^{-1} following rewetting is lower than usually reported values for whole-ecosystem fluxes (e.g., Höper et al., 2008), highlighting the importance of vegetation for belowground C accumulation and therewith GHG mitigation. However, even small differences in mean WTD (Moore and Dalva, 1993) in correlation with inter-annual variability in weather conditions (Aslan-Sungur et al., 2016; Mikhaylov et al., 2019), differing vegetation cover (Couwenberg et al., 2011), and time since rewetting (Wilson et al., 2016) pose difficulties for robust comparisons between different studies, in particular those covering larger WTD intervals. A large number of previous studies, though mainly on vegetated peatland ecosystems, are in line with our observation: although rewetting did not lead to a re-establishment of a carbon-sink function, the ecosystems were clearly characterised by significant emission reductions as compared to the state prior to rewetting (e.g. Strack and Zuback, 2013; Günther et al., 2015; Renou-Wilson et al., 2016; Franz et al., 2016).

Although the presented results have restricted validity in the context of ecosystem-scale emission magnitudes, our study design allowed to accurately compare flux magnitudes from R_h for different Danish peatland sites in the year following rewetting. Here we were able to demonstrate a GHG reduction efficiency of −33 % for fen peatlands and − 23 % for bogs. Interestingly, independent of the average emission magnitudes for the drained state, the emission factors for all sites were within a similar range when rewetted.

Due to the lack of plants in this study and the associated lack of C allocation to belowground biomass parts, naturally, no CO_2 sink function was re-established following rewetting. At the same time though, the lack of aerenchymatous plants on the rewetted mesocosms might partly explain the low observed CH_4 emissions, since vegetation can form a major pathway for methane emissions due to plant-mediated CH_4 transport, allowing methane to bypass the oxidative zone in unsaturated peat substrate (van den Berg et al., 2020). In addition, Galand et al. (2005) has highlighted the importance of peatland-vegetation for the supply of substrate, like acetate, for methanogenesis. On the other hand, *Sphagnum* peat mosses are linked to methanotrophy and CH_4 reduction on fully-saturated peatlands (Kox et al., 2021). Considering this, there is a high potential that the sites assessed in this study might show differing magnitudes of GHG under rewetting in an ecosystem-setting including the presence of typical wetland vegetation (e.g., *Typha* spp., *Phragmites* spp., *Sphagnum* spp.) in restoration or paludiculture practices.

4.2. Soil nutrient stocks and acidity explain part of the variation in emissions

The large differences observed between annual CO_2 and CH_4 emissions between sites, but also for the replicates per site, might be linked to differences regarding their drainage and land-use history, consequently affecting peat substrate quality also on a micro-scale. This was pronounced for the assessed bog sites, where sphagnum moss plant litter was the main peat building compound, where differences between replicates also have been observed regarding the von Post scale. Knowing that plant litter quality affects magnitudes of GHG emissions (Kreyling et al., 2021), numerous studies have demonstrated the high recalcitrance of sphagnum mosses and sphagnum peat (Hobbie et al., 2000; Straková et al., 2011; Kasimir et al., 2021) which are structurally more complex and hence require higher energy demands of decomposing microorganisms (Sjögersten et al., 2016). We hence hypothesise that the biochemistry of plant litter might explain differences in soil respiration for B_{SV} and B_{LV} in the drained and rewetted treatments.

In our study, we focused on simple geochemical indicators for correlations with emissions. Our findings highlighted that, for CO_2, in particular the nutrient contents of P, N, and K, frequently applied to agriculturally managed peatlands by fertilisation, and as indirectly enriched as a result of peat oxidation, as well as the soil pH, frequently affected as a result of liming, were main variables of significance. Our results show, that, for drained soils, an increase in pH above 5.5 is accompanied by an increase in CO_2 emission magnitudes, regardless of nutrient availability, which is in line with a study by Urbanová and Bárta (2020), highlighting the correlation of peat acidity and low emissions. However, under wet conditions we observed a contrary pattern, where a higher pH was correlated with lower fluxes of R_h. This is contrary to findings by Ye et al. (2012) who found an increase in CO_2 production with increasing pH also under anaerobic conditions. However, for our sites, P availability played a major role, with pH controlling the P utilisation efficiency (Luo et al., 2021), implying that with increasing pH a higher P content is required to increase respiration activity. A similar statement has been made by Ye et al. (2012), where respiratory activity was inhibited in correlation with humic substances besides pH. The overall importance of nutrients for SOM decomposition, and hence respiration rates, has been highlighted in previous studies on bog peatlands (e.g. Larmola et al., 2013; Pinsonneault et al., 2016) and numerous studies found an increase in soil respiration with higher P availability (e.g. Brake et al., 1999; Säurich et al., 2019). For TN content, we did not find clear correlations to CO_2 emissions on neither the drained nor the rewetted peatlands. In this context, we assign the slightly significant ($p < 0.05$) effect of TN on respiration rates from the drained treatment, as detected by the GAM analysis, to the general difference of emission magnitudes for the distinction between nutrient rich fen peatlands, and the nutrient poorer bogs.

In recent research (e.g., Knorr et al., 2009; Deng et al., 2017; de Jong et al., 2020), a variety of inhibitory effects were highlighted to potentially affect methanogenesis, mainly the presence of alternative electron acceptors (e.g. nitrate, ferric iron, sulfate), which are thermodynamically favoured in anaerobic respiration (Zhang et al., 2021). For instance, for B_{LV}, we observed the lowest annual CH_4 emissions of all sites, accounting for only 0.2 t CO_2eq ha^{-1} yr^{-1}. Particular for B_{LV}, as compared to B_{SV}, is a higher content of Fe in the top 25 cm of the soil column, potentially indicating the presence of Fe(III) as a preferred electron acceptor, inhibiting optimal methanogenic conditions. Van Diggelen et al. (2020) highlighted that rewetting of formerly drained coastal wetlands, such as B_{LV}, will lead to enhanced sulfate and ferric oxyhydroxide reduction rates, the latter historically known to be found on B_{LV} as “bog ore”, initially lowering CH_4 production. Further, bulk density, a soil property previously negatively correlated with methane production (Putkinen et al., 2018), was in the top 25 cm, with on average 0.13 g cm^{-3}, twice as high as compared to B_{SV}.

We found the highest overall GHG reductions following rewetting of soil from F_{SE} with an average reduction rate of 8.1 t CO_2eq ha^{-1} yr^{-1}. F_{SE} is characterised with the highest pH of all assessed sites as well as with the highest content of S in soil. In addition, as observed for all three fen peatlands, high amounts of Fe were prevailing on these sites. Interestingly, we found that higher GHG reductions, based on the mitigation of CO_2 emissions, were obtained on sites rich in Fe, while we assume that the high contents of S might have indicated the availability of SO_4^{2-} and therewith the potential for inhibited CH_4 production (Van Diggelen et al., 2020; de Jong et al., 2020) on F_{SE} and $F_Ø$ under anaerobic conditions. However, there is no consensus yet on whether iron availability in soils is correlated with GHG mitigation potentials. For instance, Chen et al. (2020), though in a study on subtropical agricultural Ultisol soils under fluctuating redox conditions, found that Fe-addition under anoxic conditions increased SOM mineralisation, and consequently CO_2 emissions. Contrary to that, Hall et al. (2016) reported a significantly decreased mineralisation of lignin methoxyl derived carbon following Fe addition, which is in line with the results by Wen et al. (2019), highlighting the role of iron in protecting peat soil organic carbon from mineralisation. However, in our study we did not assess how the iron redox cycle between the ferric and ferrous state was affected by time since rewetting. Emsens et al. (2016) proposed a higher effectiveness of rewetting for Fe-poor fen peatlands, due to lower decomposition rates of organic matter, related to anaerobic iron reduction. However, since P, found as positively affecting GHG emissions in our study, but also S, can be bound to Fe-compounds (Smolders et al., 2006; Geurts et al., 2008; Zak et al., 2010), we hypothesise that a potential trade-off between iron-reduction induced CO_2 emissions from previously stabilised SOM and iron-prevented mobilisation of P, lowering CO_2 emissions, was balanced on our assessed fens. Nonetheless, an assessment of the potential influence of alternative electron acceptors on emissions of CH_4 and CO_2 was beyond the scope of this study, focussing on simple geochemical variables, and the hypotheses regarding inhibitory effects need to be confirmed in future studies.

4.3. Implications for future management of agricultural peatlands

Peatland rewetting measures for climate and biodiversity considerations are a hot topic on global, regional, and national scales. However, cost-related hurdles for implementation (Artz et al., 2018) and the question of landownership and technical feasibility are likely to hamper current visions of meeting the climate components of legally binding international treaties by peatland rewetting. In addition, uncertainties regarding the effect of rewetting degraded peatlands on GHG balances remain.

Hence, in order to maximise the mitigation efficiency of adverse land-atmosphere carbon exchange from agriculturally used peatlands, it is critical to focus rewetting initiatives not only on areas with ideal hydrological infrastructure (Grygoruk et al., 2015; Stachowicz et al., 2022) but also on “hotspot” areas with the highest expected mitigation potential as affected by soil properties. In this context, our results present both an opportunity and a challenge. Across sites and peatland types, we were able to demonstrate a GHG reduction from heterotrophic soil respiration of approximately 28 % following rewetting, not taking biomass C budgets into account. On the other hand, independent on the initial cumulative GHG emissions in the drained state, all sites had a similar GWP following rewetting of between 8.7 and 12.6 t CO_2eq ha^{-1} yr^{-1}.

Across peat soil types, we found that the mitigation potential of CO_2 by rewetting was highest on sites that are rich in nutrients, iron and sulphur, in particular, if characterised by a near-neutral pH as well as a higher state of peat degradation. Thus, we found a higher GHG mitigation efficiency for the assessed fen peatlands. However, more similarly controlled studies on a larger variety of peatland sites are needed to make more robust predictions regarding the site-specific drivers for GHG emissions from soil respiration and their mitigation potential by rewetting. These studies will hopefully be able to further define how contents of soil nutrients, iron, sulphur, and peat acidity, as our study indicated, can be used as robust proxies for GHG mitigation efficiencies.

5. Conclusion

In our study, we focused on simple geochemical indicators for correlations with emission magnitudes. Our findings highlighted that, for CO_2, in particular the nutrient contents of P, N, and to a lesser extent K, frequently applied on agriculturally managed peatlands by fertilisation, as well as the soil pH were main variables of significance. In conclusion, overall and equal for drained and rewetted treatments, the annual cumulative GWP of the carbon-rich sites were controlled by CO_2 emissions, with CH_4 having a variable, but minor contribution and N_2O emissions being negligible. Across peat soil types, we found an indication that nutrient status, peat acidity, as well as contents of iron and sulphur, indicating the potential availability of electron acceptors, might be used as proxies for the GHG mitigation efficiency by peatland rewetting. Thus, we found an indication for prioritising drained fen peatlands for rewetting measures if quick emission reductions are required in the context of the Paris Agreement. However, more solid quantification of the expected effects, in particular when considering whole-ecosystem emission dynamics, will require more data from a larger representation of site-specific peatland biogeochemistry.

CRediT authorship contribution statement

CN developed and performed the study design and experimental work, the analysis of the data, and the writing of the manuscript. All authors contributed to the study design and the writing and reading of the manuscript and approved the final manuscript.

Data availability

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Declaration of competing interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

Acknowledgements

The authors want to thank the technical and laboratory staff at the Department of Agroecology, and in particular Michael Koppelgaard and Bodil Stensgaard, for excellent support during stages of study set-up as well as laboratory analyses. Further, the authors want to thank the reviewers for excellent suggestions regarding the improvement of the manuscript.

Funding

This study was financially supported by the PEATWISE project (https://www.eragas.eu/en/eragas/Research-projects/PEATWISE.htm) in the frame of the ERA-NET FACCE ERA-GAS. FACCE ERA-GAS received funding from the European Union's Horizon 2020 research and innovation program under the grant agreement no. 696356. This publication further was funded by the Interreg project CANAPE under the North Sea Region Programme and the European Regional Development Fund. In addition, the study was partly supported by the Aarhus University Centre for Circular Bioeconomy (CBIO, https://cbio.au.dk/en/). Further, CN received funding by the European Union's Horizon Europe programme (WET HORIZONS, grant agreement no. 101056848).

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2023.165083.

References

Artz, R. R., Faccioli, M., Roberts, M., Anderson, R. (2018). Peatland restoration–a comparative analysis of the costs and merits of different restoration methods. *CXC report, march*. Available online: https://www.climatexchange.org.uk/media/3141/peatland-restoration-methods-a-comparative-analysis.pdf, accessed: 28.03.2023.

Aslan-Sungur, G., Lee, X., Evrendilek, F., & Karakaya, N. (2016). Large interannual variability in net ecosystem carbon dioxide exchange of a disturbed temperate peatland. Sci. Total Environ., 554-555, 192–202. https://doi: https://doi.org/10.1016/j.scitotenv.2016.02.153.

Barel, J.M., Moulia, V., Hamard, S., Sytiuk, A., Jassey, V.E.J., 2021. Come rain, come Shine: peatland carbon dynamics shift under extreme precipitation. *Frontiers in Environmental*. Science 9.. https://doi:10.3389/fenvs.2021.659953.

Bonn, A., Allott, T., Evans, M., Joosten, H., Stoneman, R., 2016. Peatland restoration and ecosystem services: nature-based solutions for societal goals. In: Bonn, A., Joosten, H., Evans, M., Stoneman, R., Allott, T. (Eds.), Peatland Restoration and Ecosystem Services: Science, Policy and Practice. Cambridge University Press, Cambridge, pp. 402–417 (978-1-107-02518-9 (ISBN).

Brake, M., Höper, H., Joergensen, R.G., 1999. Land use-induced changes in activity and biomass of microorganisms in raised bog peats at different depths. Soil Biol. Biochem. 31 (11), 1489–1497. https://doi.org/10.1016/S0038-0717(99)00053-X.

Chen, C., Hall, S. J., Coward, E., & Thompson, A. (2020). Iron-mediated organic matter decomposition in humid soils can counteract protection. Nat. Commun., 11(1), 2255. https://doi: https://doi.org/10.1038/s41467-020-16071-5.

Clark, L., Strachan, I.B., Strack, M., Roulet, N.T., Knorr, K.H., Teickner, H., 2023. Duration of extraction determines CO 2 and CH 4 emissions from an actively extracted peatland in eastern Quebec. Canada. Biogeosciences 20 (3), 737–751. https://doi.org/10.5194/bg-20-737-2023.

Couwenberg, J., Thiele, A., Tanneberger, F., Augustin, J., Bärisch, S., Dubovik, D., . . . Joosten, H. (2011). Assessing greenhouse gas emissions from peatlands using vegetation as a proxy. Hydrobiologia, 674(1), 67–89. https://doi: https://doi.org/10.1007/s10750-011-0729-x.

de Jong, A.E.E., Guererro-Cruz, S., van Diggelen, J.M.H., Vaksmaa, A., Lamers, L.P.M., Jetten, M.S.M., Rasigraf, O., 2020. Changes in microbial community composition, activity, and greenhouse gas production upon inundation of drained iron-rich peat soils. Soil Biol. Biochem. 149.. https://doi:10.1016/j.soilbio.2020.107862.

Deng, J., McCalley, C.K., Frolking, S., Chanton, J., Crill, P., Varner, R., Li, C., 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.

Dezzeo, N., Grandez-Rios, J., Martius, C., et al., 2021. Degradation-driven changes in fine root carbon stocks, productivity, mortality, and decomposition rates in a palm swamp peat forest of the Peruvian Amazon. Carbon Balance Manage 16, 33. https://doi.org/10.1186/s13021-021-00197-0.

Elsgaard, L., Görres, C.-M., Hoffmann, C.C., Blicher-Mathiesen, G., Schelde, K., Petersen, S.O., 2012. Net ecosystem exchange of CO2 and carbon balance for eight temperate organic soils under agricultural management. Agric. Ecosyst. Environ. 162, 52–67. https://doi.org/10.1016/j.agee.2012.09.001.

Emsens, W.-J., Aggenbach, C. J. S., Schoutens, K., Smolders, A. J. P., Zak, D., & van Diggelen, R. (2016). Soil Iron content as a predictor of carbon and nutrient mobilization in rewetted fens. PLoS One, 11(4), e0153166-e0153166. https://doi: https://doi.org/10.1371/journal.pone.0153166.

Evans, C. D., Peacock, M., Baird, A. J., Artz, R. R. E., Burden, A., Callaghan, N., . . . Morrison, R. (2021). Overriding water table control on managed peatland greenhouse gas emissions. Nature, 593(7860), 548–552. https://doi: https://doi.org/10.1038/s41586-021-03523-1.

Franz, D., Koebsch, F., Larmanou, E., Augustin, J., & Sachs, T. (2016). High net CO2 and CH4 release at a eutrophic shallow lake on a formerly drained fen. Biogeosciences, 13(10), 3051–3070. https://doi: https://doi.org/10.5194/bg-13-3051-2016.

Galand, P. E., Fritze, H., Conrad, R., & Yrjala, K. (2005). Pathways for methanogenesis and diversity of methanogenic archaea in three boreal peatland ecosystems. Appl. Environ. Microbiol., 71(4), 2195–2198. https://doi: https://doi.org/10.1128/AEM.71.4.2195-2198.2005.

Geurts, J.J.M., Smolders, A.J.P., Verhoeven, J.T.A., Roelofs, J.G.M., Lamers, L.P.M., 2008. Sediment Fe:PO4 ratio as a diagnostic and prognostic tool for the restoration of macrophyte biodiversity in fen waters. Freshw. Biol. 53 (10), 2101–2116. https://doi.org/10.1111/j.1365-2427.2008.02038.x.

Greve, M. H., Greve, M. B., Peng, Y., Pedersen, B. F., Møller, A. B., Lærke, P. E., et al. (2021). *Vidensyntese Om Kulstofrig Lavbundsjord*. Tjele, Denmark: *Aarhus University, DCA Rapport* 137. Available online: https://pure.au.dk/portal/files/214394346/Vidensyntese_kulstofrig_lavbundsjord_3003_2021_rev.pdf, accessed: 04.03.2022.

Grygoruk, M., Bańkowska, A., Jabłońska, E., Janauer, G.A., Kubrak, J., Mirosław-Świątek, D., Kotowski, W., 2015. Assessing habitat exposure to eutrophication in restored wetlands: model-supported ex-ante approach to rewetting drained mires. J. Environ. Manag. 152, 230–240. https://doi.org/10.1016/j.jenvman.2015.01.049.

Günther, A., Huth, V., Jurasinski, G., & Glatzel, S. (2015). The effect of biomass harvesting on greenhouse gas emissions from a rewetted temperate fen. GCB Bioenergy, 7(5), 1092–1106. https://doi: https://doi.org/10.1111/gcbb.12214.

Hall, S.J., Silver, W.L., Timokhin, V.I., Hammel, K.E., 2016. Iron addition to soil specifically stabilized lignin. Soil Biol. Biochem. 98, 95–98. https://doi.org/10.1016/j.soilbio.2016.04.010.

Hergoualc'h, K., Hendry, D.T., Murdiyarso, D., et al., 2017. Total and heterotrophic soil respiration in a swamp forest and oil palm plantations on peat in Central Kalimantan, Indonesia. Biogeochemistry 135, 203–220. https://doi.org/10.1007/s10533-017-0363-4.

Hobbie, S.E., Schimel, J.P., Trumbore, S.E., Randerson, J.R., 2000. Controls over carbon storage and turnover in high-latitude soils. Glob. Chang. Biol. 6 (S1), 196–210. https://doi.org/10.1046/j.1365-2486.2000.06021.x.

Höper, H., Augustin, J., Cagampan, J., Drösler, M., Lundin, L., Moors, E., . . . Wilson, D. (2008). Restoration of peatlands and greenhouse gas balances. In: *Peatlands and Climate Change*, International Peat Society-ISBN 9789529940110- p. 182-210.

Hu, Z., Liu, S., Liu, X., et al., 2016. Soil respiration and its environmental response varies by day/night and by growing/dormant season in a subalpine forest. Sci. Rep. 6, 37864. https://doi.org/10.1038/srep37864.

Huth, V., Günther, A., Jurasinski, G., Glatzel, S., 2013. The effect of an exceptionally wet summer on methane effluxes from a 15-year re-wetted fen in north-East Germany. Mires and Peat 13 (02), 1–7.

Huth, V., Günther, A., Bartel, A., Hofer, B., Jacobs, O., Jantz, N., Jurasinski, G., 2020. Topsoil removal reduced in-situ methane emissions in a temperate rewetted bog grassland by a hundredfold. Sci. Total Environ. 721.. https://doi:10.1016/j.scitotenv.2020.137763.

Hütsch, B.W., 2001. Methane oxidation, nitrification, and counts of methanotrophic bacteria in soils from a long-term fertilization experiment ("Ewiger Roggenbau" at Halle). J. Plant Nutr. Soil Sci. 164 (1), 21–28. https://doi.org/10.1002/1522-2624(200102)164:1<21::AID-JPLN21>3.0.CO;2-B.

IPCC (2014). 2013 supplement to the 2006 IPCC guidelines for National Greenhouse gas Inventories: wetlands, Hiraishi, T., Krug, T., Tanabe, K., Srivastava, N., Baasansuren, J., Fukuda, M. and Troxler, T.G. (eds). Published: *IPCC, Switzerland*. Available online:

https://www.ipcc.ch/publication/2013-supplement-to-the-2006-ipcc-guidelines-for-national-greenhouse-gas-inventories-wetlands/, accessed: 17.02.2022.

Iversen, C.M., Childs, J., Norby, R.J., Ontl, T.A., Kolka, R.K., Brice, D.J., McFarlane, K.J., Hanson, P.J., 2018. Fine-root growth in a forested bog is seasonally dynamic, but shallowly distributed in nutrient-poor peat. Plant Soil 424, 123–143. https://doi.org/10.1007/s11104-017-3231-z.

Järveoja, J., Nilsson, M.B., Crill, P.M., et al., 2020. Bimodal diel pattern in peatland ecosystem respiration rebuts uniform temperature response. Nat. Commun. 11, 4255. https://doi.org/10.1038/s41467-020-18027-1.

Joosten, H., 2016. Peatlands across the globe. In: Bonn, A., Joosten, H., Evans, M., Stoneman, R., Allott, T. (Eds.), Peatland Restoration and Ecosystem Services: Science, Policy and Practice. Cambridge University Press, Cambridge, pp. 402–417 (978-1-107-02518-9 (ISBN).

Joosten, H., Clarke, D., 2002. Wise Use of Mires and Peatlands: Background and Principles Including a Framework for Decision-Making. International mire conservation group 951-97744-8-3.

Jovani-Sancho, A.J., Cummins, T. & Byrne, K.A. (2018) Soil respiration partitioning in afforested temperate peatlands. Biogeochemistry 141, 1–21. doi:10.1007/s10533-018-0496-0

Jurasinski, G., Koebsch, F., Guenther, A., & Beetz, S. (2014). flux: Flux rate calculation from dynamic closed chamber measurements. R package version 0.3-0. https://CRAN.R-project.org/package=flux.

Karki, S., Elsgaard, L., Audet, J., & Lærke, P. E. (2014). Mitigation of greenhouse gas emissions from reed canary grass in paludiculture: effect of groundwater level. Plant Soil, 383(1), 217–230. https://doi: https://doi.org/10.1007/s11104-014-2164-z.

Karki, S., Elsgaard, L., & Lærke, P. E. (2015). Effect of reed canary grass cultivation on greenhouse gas emission from peat soil at controlled rewetting. Biogeosciences, 12(2), 595–606. https://doi: https://doi.org/10.5194/bg-12-595-2015.

Kasimir, Å., He, H., Jansson, P.-E., Lohila, A., Minkkinen, K., 2021. Mosses are important for soil carbon sequestration in forested peatlands. *Frontiers in Environmental*. Science 9.. https://doi:10.3389/fenvs.2021.680430.

Kleber, M., 2010. What is recalcitrant soil organic matter. Environ. Chem. 7 (4), 320–332. https://doi.org/10.1071/EN10006.

Knorr, K.-H., Lischeid, G., Blodau, C., 2009. Dynamics of redox processes in a minerotrophic fen exposed to a water table manipulation. Geoderma 153 (3), 379–392. https://doi.org/10.1016/j.geoderma.2009.08.023.

Knox, S. H., Jackson, R. B., Poulter, B., McNicol, G., Fluet-Chouinard, E., Zhang, Z., . . . Zona, D. (2019). FLUXNET-CH4 synthesis activity: objectives, observations, and future directions. Bull. Am. Meteorol. Soc., 100(12), 2607–2632. https://doi: https://doi.org/10.1175/bams-d-18-0268.1.

Komulainen, V.-M., Tuittila, E.-S., Vasander, H., & Laine, J. (1999). Restoration of drained peatlands in southern Finland: initial effects on vegetation change and CO2 balance. J. Appl. Ecol., 36(5), 634–648. https://doi: https://doi.org/10.1046/j.1365-2664.1999.00430.x.

Kox, M., Smolders, A., Speth, D., Lamers, L., Op den Camp, H., Jetten, M., van Kessel, M. (2021) A novel laboratory-scale Mesocosm setup to study methane emission mitigation by Sphagnum mosses and associated Methanotrophs. Front. Microbiol. 12:652486. https://doi: https://doi.org/10.3389/fmicb.2021.651103.

Kreyling, J., Tanneberger, F., Jansen, F., van der Linden, S., Aggenbach, C., Blüml, V., . . . Jurasinski, G. (2021). Rewetting does not return drained fen peatlands to their old selves. Nat. Commun., 12(1), 5693. https://doi: https://doi.org/10.1038/s41467-021-25619-y.

Larmola, T., Tuittila, E., Tiirola, M., Nykänen, H., Martikainen, P.J., Yrjälä, K., Tuomivirta, T., Fritze, H., 2010. The role of Sphagnum mosses in the methane cycling of a boreal mire. Ecology 91, 2356–2365. https://doi.org/10.1890/09-1343.1.

Larmola, T., Bubier, J.L., Kobyljanec, C., Basiliko, N., Juutinen, S., Humphreys, E., Moore, T.R., 2013. Vegetation feedbacks of nutrient addition lead to a weaker carbon sink in an ombrotrophic bog. Glob. Chang. Biol. 19 (12), 3729–3739. https://doi.org/10.1111/gcb.12328.

Limpens, J., Berendse, F., Blodau, C., Canadell, J. G., Freeman, C., Holden, J., . . . Schaepman-Strub, G. (2008). Peatlands and the carbon cycle: from local processes to global implications – a synthesis. Biogeosciences, 5(5), 1475–1491. https://doi: https://doi.org/10.5194/bg-5-1475-2008.

Liobikienė, G., Butkus, M., 2017. The European Union possibilities to achieve targets of Europe 2020 and Paris agreement climate policy. Renew. Energy 106 (C), 298–309 Retrieved from https://EconPapers.repec.org/RePEc:eee:renene:v:106:y:2017:i:c:p:298-309 Retrieved from.

Liu, N., Jørgensen, U., & Lærke, P. E. (2013). Quality determination of biomass for combustion: a new high-throughput microwave digestion method prior to elemental analysis by inductively coupled plasma–optical emission spectroscopy. Energy Fuel, 27(12), 7485–7488. https://doi: https://doi.org/10.1021/ef4016747.

Lloyd, C. R., Rebelo, L.-M., & Finlayson, C. M. (2013). Providing low-budget estimations of carbon sequestration and greenhouse gas emissions in agricultural wetlands. Environ. Res. Lett., 8(1), 015010. https://doi: https://doi.org/10.1088/1748-9326/8/1/015010.

Lloyd, J., Taylor, J. A. (1994). On the temperature dependence of soil respiration. Funct. Ecol., 8(3), 315–323. https://doi: https://doi.org/10.2307/2389824.

Loisel, J., Yu, Z., Beilman, D. W., Camill, P., Alm, J., Amesbury, M. J., . . . Zhou, W. (2014). A database and synthesis of northern peatland soil properties and Holocene carbon and nitrogen accumulation. The Holocene, 24(9), 1028–1042. https://doi: https://doi.org/10.1177/0959683614538073.

Luo, T., Zhu, Y., Lu, W., Chen, L., Min, T., Li, J., Wei, C., 2021. Acidic compost tea enhances phosphorus availability and cotton yield in calcareous soils by decreasing soil pH. Acta Agric. Scand. Sect. B — Soil Plant Sci. 71 (8), 657–666. https://doi.org/10.1080/09064710.2021.1933161.

Maljanen, M., Sigurdsson, B. D., Guðmundsson, J., Óskarsson, H., Huttunen, J. T., & Martikainen, P. J. (2010). Greenhouse gas balances of managed peatlands in the Nordic countries – present knowledge and gaps. Biogeosciences, 7(9), 2711–2738. https://doi: https://doi.org/10.5194/bg-7-2711-2010.

Marra, G., Wood, S.N., 2011. Practical variable selection for generalized additive models. Computational Statistics & Data Analysis 55 (7), 2372–2387. https://doi.org/10.1016/j.csda.2011.02.004.

Mikhaylov, O., Zagirova, S., & Miglovets, M. (2019). Seasonal and inter-annual variability of carbon dioxide exchange at a boreal peatland in north-east European Russia. Mires and Peat, 24, 1–16. https://doi: 10.19189/MaP.2017.OMB.293.

Mitsch, W.J., Gosselink, J.G., 2000. The value of wetlands: importance of scale and landscape setting. Ecol. Econ. 35 (1), 25–33. https://doi.org/10.1016/S0921-8009(00)00165-8.

Moore, T. R., & Dalva, M. (1993). The influence of temperature and water table position on carbon dioxide and methane emissions from laboratory columns of peatland soils. J. Soil Sci., 44(4), 651–664. https://doi: https://doi.org/10.1111/j.1365-2389.1993.tb02330.x.

Myhre, G., Shindell, D., Bréon, F. M., Collins, W., Fuglestvedt, J., Huang, J., . . . Zhang, H. (Eds.). (2013). Anthropogenic and Natural Radiative Forcing. In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge, UK: Cambridge University Press., ISBN 9781107057991.

Parish, F., Sirin, A., Charman, D., Joosten, H., Minayeva, T., Silvius, M. and Stringer, L. (Eds.) 2008. Assessment on Peatlands, Biodiversity and Climate Change: Main Report. Global Environment Centre, Kuala Lumpur and Wetlands International, Wageningen. ISBN: 978–983–43751-0-2.

Petersen, S. O., Hoffmann, C. C., Schäfer, C. M., Blicher-Mathiesen, G., Elsgaard, L., Kristensen, K., . . . Greve, M. H. (2012). Annual emissions of CH4 and N2O, and ecosystem respiration, from eight organic soils in Western Denmark managed by agriculture. Biogeosciences, 9(1), 403–422. https://doi: https://doi.org/10.5194/bg-9-403-2012.

Pinsonneault, A. J., Moore, T. R., & Roulet, N. T. (2016). Effects of long-term fertilization on peat stoichiometry and associated microbial enzyme activity in an ombrotrophic bog. Biogeochemistry, 129(1), 149–164. https://doi: https://doi.org/10.1007/s10533-016-0224-6.

Putkinen, A., Tuittila, E.-S., Siljanen, H.M.P., Bodrossy, L., Fritze, H., 2018. Recovery of methane turnover and the associated microbial communities in restored cutover peatlands is strongly linked with increasing Sphagnum abundance. Soil Biol. Biochem. 116, 110–119. https://doi.org/10.1016/j.soilbio.2017.10.005.

R Core Team, 2020. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/.

Saidy, A. R., Razie, F., Aidawati, N., & Hidayat, T. (2020). Increases in greenhouse gases following the use of peatlands for agricultural areas. IOP Conference Series: Earth and Environmental Science, 499, 012021. https://doi: https://doi.org/10.1088/1755-1315/499/1/012021.

Säurich, A., Tiemeyer, B., Don, A., Fiedler, S., Bechtold, M., Amelung, W., Freibauer, A., 2019. Drained organic soils under agriculture — the more degraded the soil the higher the specific basal respiration. Geoderma 355, 113911. https://doi.org/10.1016/j.geoderma.2019.113911.

Shurpali, N.J., Hyvönen, N.P., Huttunen, J.T., Biasi, C., Nykänen, H., Pekkarinen, N., Martikainen, P.J., 2008. Bare soil and reed canary grass ecosystem respiration in peat extraction sites in eastern Finland. Tellus B: Chemical and Physical Meteorology 60 (2), 200–209. https://doi.org/10.1111/j.1600-0889.2007.00325.x.

Sjögersten, S., Caul, S., Daniell, T.J., Jurd, A.P.S., O'Sullivan, O.S., Stapleton, C.S., Titman, J.J., 2016. Organic matter chemistry controls greenhouse gas emissions from permafrost peatlands. Soil Biol. Biochem. 98, 42–53. https://doi.org/10.1016/j.soilbio.2016.03.016.

Smolders, A. J. P., Lamers, L. P. M., Lucassen, E. C. H. E. T., Van Der Velde, G., & Roelofs, J. G. M. (2006). Internal eutrophication: how it works and what to do about it—a review. Chem. Ecol., 22(2), 93–111. https://doi: https://doi.org/10.1080/02757540600579730.

Stachowicz, M., Manton, M., Abramchuk, M., Banaszuk, P., Jarašius, L., Kamocki, A., Povilaitis, A., Samerkhanova, A., Schäfer, A., Sendžikaitė, J., Wichtmann, W., 2022. To store or to drain—to lose or to gain? Rewetting drained peatlands as a measure for increasing water storage in the transboundary Neman River basin. Sci. Total Environ. 829, 154560. https://doi.org/10.1016/j.scitotenv.2022.154560.

Strack, M., & Zuback, Y. C. A. (2013). Annual carbon balance of a peatland 10 yr following restoration. Biogeosciences, 10(5), 2885–2896. https://doi: https://doi.org/10.5194/bg-10-2885-2013.

Straková, P., Niemi, R. M., Freeman, C., Peltoniemi, K., Toberman, H., Heiskanen, I., . . . Laiho, R. (2011). Litter type affects the activity of aerobic decomposers in a boreal peatland more than site nutrient and water table regimes. Biogeosciences, 8(9), 2741–2755. https://doi: https://doi.org/10.5194/bg-8-2741-2011.

Tiemeyer, B., Freibauer, A., Borraz, E.A., Augustin, J., Bechtold, M., Beetz, S., Drösler, M., 2020. A new methodology for organic soils in national greenhouse gas inventories: data synthesis, derivation and application. Ecol. Indic. 109, 105838. https://doi.org/10.1016/j.ecolind.2019.105838.

Urbanová, Z., Bárta, J., 2020. Recovery of methanogenic community and its activity in long-term drained peatlands after rewetting. Ecol. Eng. 150.. https://doi:10.1016/j.ecoleng.2020.105852.

van den Berg, M., van den Elzen, E., Ingwersen, J., et al., 2020. Contribution of plant-induced pressurized flow to CH4 emission from a Phragmites fen. Sci. Rep. 10, 12304. https://doi.org/10.1038/s41598-020-69034-7.

Van Diggelen, J., Lamers, L., Loermans, J., Rip, W., & Smolders, A. (2020). Towards more sustainable hydrological management and land use of drained coastal peatlands-a biogeochemical balancing act. Mires and Peat, 26, 17, 12pp. https://doi:10.19189/MaP.2019.APG.StA.1771

von Post, L., 1922. Sveriges geologiska undersöknings torvinventering och några av dess hittills vunna resultat.

von Post, L., Granlund, E., 1926. Södra Sveriges Torvtillgångar: Norstedt.

Vybornova, O., van Asperen, H., Pfeiffer, E., & Kutzbach, L. (2019). High N 2 O and CO 2 emissions from bare peat dams reduce the climate mitigation potential of bog rewetting practices. Mires and Peat, 24(04), 1–22. https://doi: 10.19189/MaP.2017.SNPG.304.

Wen, Y., Zang, H., Ma, Q., Evans, C. D., Chadwick, D. R., & Jones, D. L. (2019). Is the 'enzyme latch' or 'iron gate' the key to protecting soil organic carbon in peatlands? Geoderma, 349, 107–113. https://doi: https://doi.org/10.1016/j.geoderma.2019.04.023.

Weslien, P., Kasimir Klemedtsson, Å., Börjesson, G., & Klemedtsson, L. (2009). Strong pH influence on N2 O and CH4 fluxes from forested organic soils. Eur. J. Soil Sci., 60(3), 311–320. https://doi: https://doi.org/10.1111/j.1365-2389.2009.01123.x.

Wigand, C., Brennan, P., Stolt, M., et al., 2009. Soil respiration rates in coastal marshes subject to increasing watershed nitrogen loads in southern New England, USA. Wetlands 29, 952–963. https://doi.org/10.1672/08-147.1.

Wilson, D., Alm, J., Laine, J., Byrne, K. A., Farrell, E. P., & Tuittila, E.-S. (2009). Rewetting of cutaway peatlands: are we re-creating hot spots of methane emissions? Restor. Ecol., 17(6), 796–806. https://doi: https://doi.org/10.1111/j.1526-100x.2008.00416.x.

Wilson, D., Blain, D., Couwenberg, J., Evans, C., Murdiyarso, D., Page, S., . . . Strack, M. (2016). Greenhouse gas emission factors associated with rewetting of organic soils. Mires and Peat, Volume 17, 04, 1–28, https://doi: 10.19189/MaP.2016.OMB.222.

Wood, S.N., 2011. Fast stable restricted maximum likelihood and marginal likelihood estimation of semiparametric generalized linear models. Journal of the Royal Statistical Society: Series B (Statistical Methodology) 73 (1), 3–36. https://doi.org/10.1111/j.1467-9868.2010.00749.x.

Wood, S. N., Pya, N., & Säfken, B. (2016). Smoothing parameter and model selection for general smooth models. J. Am. Stat. Assoc., 111(516), 1548–1563. https://doi: https://doi.org/10.1080/01621459.2016.1180986.

Ye, R., Jin, Q., Bohannan, B., Keller, J.K., McAllister, S.A., Bridgham, S.D., 2012. pH controls over anaerobic carbon mineralization, the efficiency of methane production, and methanogenic pathways in peatlands across an ombrotrophic–minerotrophic gradient. Soil Biol. Biochem. 54, 36–47. https://doi.org/10.1016/j.soilbio.2012.05.015.

Yu, Z., Loisel, J., Brosseau, D. P., Beilman, D. W., & Hunt, S. J. (2010). Global peatland dynamics since the last glacial maximum. Geophys. Res. Lett., 37(13), https://doi: https://doi.org/10.1029/2010gl043584.

Zak, D., Wagner, C., Payer, B., Augustin, J., & Gelbrecht, J. (2010). Phosphorus mobilization in rewetted fens: the effect of altered peat properties and implications for their restoration. Ecol. Appl., 20(5), 1336–1349. Retrieved from http://www.jstor.org/stable/25680382.

Zhang, H., Tuittila, E.-S., Korrensalo, A., Laine, A.M., Uljas, S., Welti, N., Lohila, A., 2021. Methane production and oxidation potentials along a fen-bog gradient from southern boreal to subarctic peatlands in Finland. Glob. Chang. Biol. 27 (18), 4449–4464. https://doi.org/10.1111/gcb.15740.