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Impact of peat depth on soil conditions, plant growth, and greenhouse gas emissions on a boreal agricultural drained peatland

Milla Niiranen[a,*], Liisa Kulmala[b,c], Miika Läpikivi[a,d], Henriikka Vekuri[b], Erkki Joki-Tokola[a], Timo Lötjönen[a], Hannu Marttila[d], Annalea Lohila[b,e], Jari Liski[b], Maarit Liimatainen[a,d]

[a] Natural Resources Institute Finland, Oulu, Finland
[b] Finnish Meteorological Institute, Climate System Research Unit, Helsinki, Finland
[c] Institute for Atmospheric and Earth System Research, Forest Sciences, University of Helsinki, Finland
[d] Water, Energy and Environmental Engineering Research Unit, University of Oulu, Finland
[e] Institute for Atmospheric and Earth System Research, Physics, University of Helsinki, Finland

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ABSTRACT

Drained agricultural peatlands are significant sources of greenhouse gas (GHG) emissions, particularly carbon dioxide (CO_2) and nitrous oxide (N_2O). Peat thickness typically varies widely between sites due to cultivation history, but its effect on soil conditions, plant growth, and GHG emissions are poorly understood and seldom considered in national GHG inventories and emission mitigation strategies. Here, we present a three-year study of a drained agricultural peatland in northern Finland, with peat depth ranging from 20 to 80 cm. We measured soil properties, plant growth and GHG fluxes, to evaluate how peat depth influences ecosystem processes and emissions. Plots with deeper peat exhibited more stable and shallow water table depth (WTD), and had higher peaks in soil nitrate concentrations, but these differences did not result in significant changes in soil moisture, plant growth, or GHG fluxes. However, we observed periods when drier conditions in mull soil limited ecosystem respiration (R_e), while WTD indicated a stronger negative effect on R_e and a positive effect on CH_4 fluxes in plots with deeper peat layer. Our results suggest that peat depth alone does not improve GHG inventories since shallow-peated drained peatlands can have as high GHG emissions as the thicker ones, and hydrological parameters should be prioritized instead. However, mitigation measures such as rewetting can still be targeted towards fields with thicker peat layer, as larger soil organic carbon stocks can be affected, and rewetting is likely easier due to more stable WTD and higher water holding capacity.

1. Introduction

Peatlands are significant carbon pools storing approximately 85% of the carbon in the Northern Hemisphere (Hugelius et al., 2020). Using peatlands for agriculture involves management practices like drainage, ploughing, fertilization, and liming, which enhance the decomposition of organic matter and turn the soils into substantial net carbon dioxide (CO_2) and nitrous oxide (N_2O) sources globally (Maljanen et al., 2007, 2010; Qiu et al., 2021; Gerin et al., 2023). In Finland, organic soils cover around 10% of the cultivated area (Myllys and Sinkkonen, 2004) but produce 60% of the total emissions reported in Land use, land-use change, and forestry (LULUCF; Statistics Finland, 2024), making them a potential target in climate change mitigation.

Cultivated fields can have organic layers ranging from deep, over-meter-thick peat to shallow, mineral-mixed soils that no longer qualify as peat (Pohjankukka et al., 2025), reflecting historical drainage and cultivation practices. According to the IPCC (2014), soils with at least a 10-cm horizon and $\geq$ 12–18% organic carbon (OC) are classified as "organic soils," while in Finland, organic soils are further divided into peat soils with at least 20% OC and mull soils with 10–19.9% OC in the plough layer (Yli-Halla et al., 2022).

Drainage causes rapid consolidation of the peat matrix, followed by gradual subsidence driven mainly by the oxidation of organic matter combined with decreased litter inputs due to harvesting (Hooijer et al., 2012). As decomposition progresses, fiber content of the peat decreases resulting in increased bulk density and higher proportion of small pores

* Corresponding author.
E-mail address: milla.niiranen@luke.fi (M. Niiranen).

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(Waddington et al., 2015). Consequently, shallow peat soils with more degraded structure can exhibit deeper and more variable water table and be more vulnerable to drought, whereas thicker peat layers with higher porosity can have greater water storage and buffering capacity under limited precipitation (Berglund and Berglund, 2011; Waddington et al., 2015; Menberu et al., 2021). Water table depth (WTD) and soil moisture strongly regulate microbial activity, nutrient cycling, and plant growth and consequently regulate the rate of GHG emissions (Guntiñas et al., 2012; Heimsch et al., 2021; Mäkelä et al., 2022). Also, as the peat gets more degraded the amount of easily degradable carbon decreases (Jayasekara et al., 2025), and the remaining organic matter in peatlands with long cultivation history could be more recalcitrant to further breakdown compared to more recently drained peatlands (Swails et al., 2018). This may constrain microbial activity and further limit GHG emissions (Berglund and Berglund, 2011).

Yet peat depth is not currently considered in national GHG emission inventories or in emission reduction strategies, and only little research has been done on the impact of peat depth on GHG emissions from cultivated peatlands. In some studies, increases in soil organic carbon (SOC) have found to contribute to higher N_2O emissions (Li et al., 2005, 2018; Ye et al., 2016; Kelley et al., 2024), while Yli-Halla et al. (2022) and Purviņa et al. (2024) found no difference in ecosystem respiration, or N_2O and CH_4 fluxes between shallow and deep peat soils. The study by Yli-Halla et al. (2022) covered, however, only 8 weeks during one summer and did not consider the effects of other environmental variables. Similarly, other studies have shown that soils with low amount of organic carbon and peat mixed with mineral soil can release as high CO_2 and N_2O emissions as deep unmixed peat soils with higher organic carbon stocks (Leiber-Sauheitl et al., 2013; Eickenscheidt et al., 2015; Jerray et al., 2024). The interactive effects of peat depth with WTD, moisture conditions, and nutrient status on GHG emissions remain unclear, yet understanding them is critical for improving current GHG inventory methods and guiding mitigation activities.

To address these gaps in previous studies, our aim was to quantify how peat depth influences soil conditions, plant growth, and GHG exchange in drained agricultural fields. Specifically, we addressed three research questions:

1: Does peat depth significantly affect soil conditions such as soil moisture, and ammonium and nitrate concentrations?

2: Does variation in peat depth affect photosynthetic potential, plant growth and yield?

3: Does peat depth cause significant variation in GHG fluxes?

For the purpose, we measured soil properties, photosynthesis, and plant growth together with GHG fluxes for three years from May 2019 to December 2021 on a drained peatland in northern Finland in a field where the depth of organic layer varies from 15 cm to 80 cm. To our knowledge, this is among the first long-term, high-frequency study focusing on peat depth effects in a cultivated northern peatland with shallow peat layers and a long cultivation history.

2. Methods

2.1. Study area

NorPeat research platform (26 ha) is located at Ruukki, Finland (25.00°E, 64.42°N, ca. 45 m above the mean sea level) and managed by Natural Resources Institute Finland (Luke). The field is a former minerotrophic peatland cultivated under grass-intensive crop rotation for beef cattle feed production, where grass is cultivated for 3–4 years before 1–2 years of cereal crops. The field has been in agricultural use for around 100 years (Yli-Halla et al., 2022) and it belongs to the European network of platforms for Analysis and Experimentation on Ecosystems (AnaEE, 2025).

The organic layer of the field ranges from approximately 20–80 cm and is divided into deep peat (>60 cm), shallow peat (30–60 cm), and mull soil (<30 cm). The surface soil is sedge peat (pH 5.8), underlain by fine sand, silt and silty clay with sulfidic subsoil (Yli-Halla et al., 2022). The field is divided into eight drainage blocks (3.0–3.8 ha, Fig. 1), with blocks 1–7 included in this study. Blocks with deep peat (1, 2, 4, part of 3) are classified as Sapric Histosols (Sulfidic) while shallow peat blocks (5–7, part of 3) are classified as Gleyic Histic Umbrisols (Sulfidic, Thionic) (IUSS Working Group WRB, 2022). Especially in blocks 5 and 6, the plough layer (0–25 cm) is currently at the boundary between mineral and organic soil due to a longer cultivation history and the oxidation of soil organic matter. Deep peat blocks have larger SOC stocks (Fig. 2; Table S1) whereas topsoil SOC concentrations (>16%) and C:N ratios (16–18:1) are relatively uniform across blocks (Fig. S1, Table S1). Bulk density at 0–20 cm ranges from 0.35 to 0.65 kg dm^{-3}, with higher values in shallower peat (Fig. S2). The field is drained using subsurface pipes at around 1 m depth. Additional information on the drainage system and soil properties of the field can be found in Yli-Halla et al. (2022).

The long-term (1991–2021) mean annual temperature of the area is 3.2 °C, with February being the coldest month (-8.3 °C) and July the warmest month (16.3 °C). Mean annual precipitation is 556 mm with July and August as the wettest months ($\geq$ 70 mm per month) and March and April as the driest months ($\leq$ 30 mm) (Finnish Meteorological Institute, 2024a). Snow cover typically lasts from late November to late April, and the growing season starts at the beginning of May and ends at the beginning of October (Kersalo and Pirinen, 2009; Finnish Meteorological Institute, 2024b). During the study period, total precipitation in 2020 was 176 mm higher than the long-term average, mainly due to 133 mm more precipitation in July. In 2019, total precipitation was 47 mm lower than the long-term average due to drier April and July (i.e. 50 mm and 30 mm less precipitation, respectively). Otherwise, precipitation was distributed quite evenly. The average annual temperature in 2020 was also 1.9 °C higher than the long-term average due to warmer winter months (Finnish Meteorological Institute, 2024a).

2.2. Management of the site during the study

In 2019 blocks 1–4 were sown with barley and timothy-meadow fescue mixture. Barley was harvested on September 25th and the grass was left to grow. Blocks 5–7 had mature timothy-meadow fescue mixture, established in 2017, for the entire monitoring period. In 2020, parts of the field were resown because the grass grew poorly after winter, especially on the recently established blocks 1–4.

All blocks were fertilized with mineral fertilizer (Table 1.). Blocks 5–7 were managed otherwise similarly but in October 2019, slurry was also applied to block 6 to see if a slurry application in autumn causes an increase in nutrient loading (Pham et al., 2023). Glyphosate was applied on September 26th 2021 to kill remaining vegetation.

Data from blocks 1–4 in 2019 were not included in this study because of differences in crop species and management. In 2020 and 2021 all blocks (1−7) were included in the analysis.

3. Measurements

3.1. Ecosystem respiration, methane and nitrous oxide emissions

Greenhouse gas emissions were measured from May 2019 to December 2021. During snow-free seasons, ecosystem respiration (R_e), methane (CH_4) and nitrous oxide (N_2O) exchange were measured with a closed static chamber method (Fig. S3). Metal collars (60 cm×60 cm) with water grooves were permanently installed in the soil at a 20 cm depth and removed during harvest. Collar placements were systematically selected and there were four replicates for each of the 7 blocks. A dark metal chamber (60 cm × 60 cm × 40 cm) with an air mixing fan was placed on top of the collar and during the 45 min closure time four gas samples (0, 15, 30, 45 min) of 20 ml were taken with a 60 ml plastic syringe and immediately injected into pre-evacuated glass vials. Extra collars were used if the vegetation was taller.

During snow-covered seasons, GHG emissions were measured using a

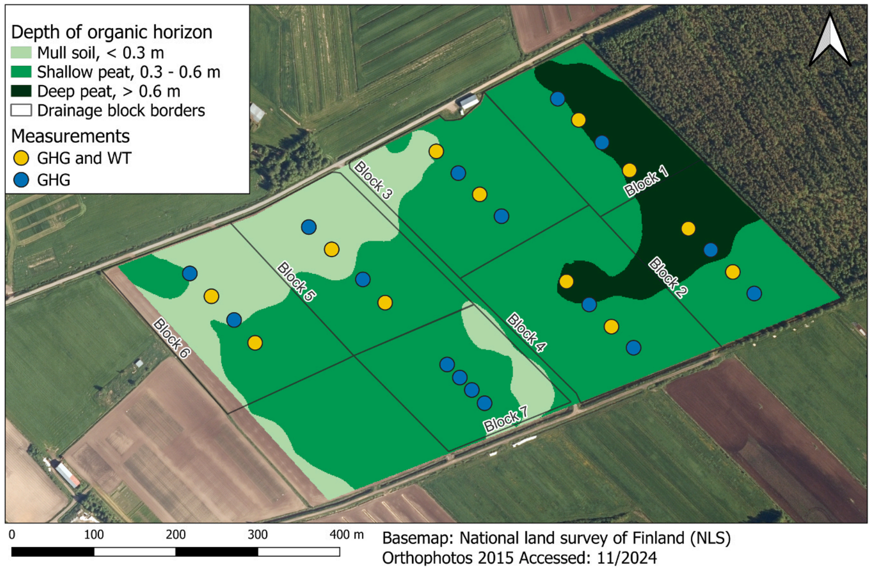

Fig. 1. Research area and monitoring layout. Distribution of measurement locations within the whole field shown with a depth of organic horizon. The organic soil layer is classified as mull (<30 cm), shallow peat (30–60 cm) and deep peat (>60 cm). Peat depth data is derived from a ground penetrating radar (GPR) survey conducted in July 2020 (Yli-Halla et al., 2022).

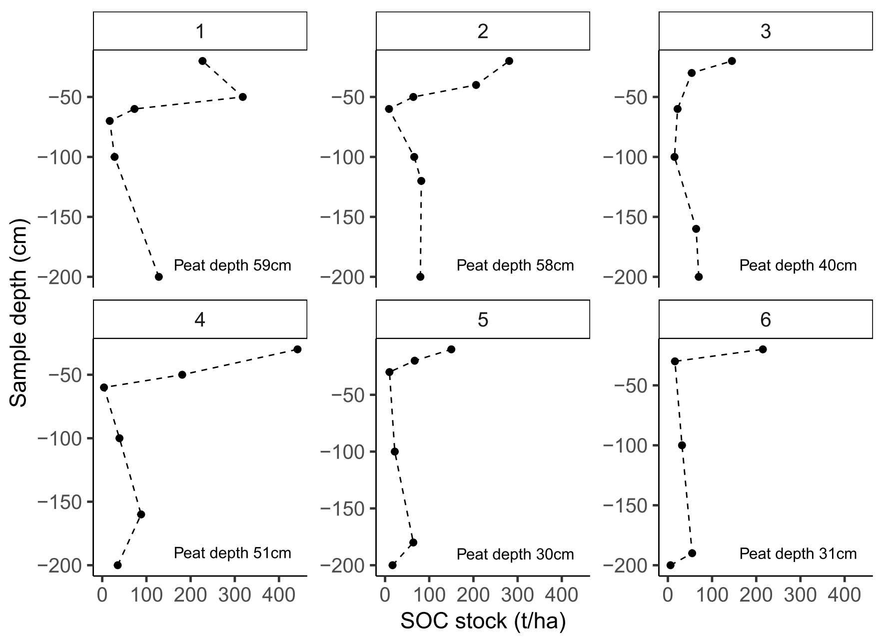

Fig. 2. Soil organic carbon (SOC) stocks (t/ha) at different sampling depths from 0 to 200 cm in blocks 1–6 and average peat depth (cm) for each block. Data are from the supplementary material of Yli-Halla et al. (2022).

Table 1
Rates (kg ha^{-1}) of nitrogen (N), phosphorus (P) and potassium (K) fertilization, harvesting and fertilization dates.

Year	Block	N (kg ha^{-1})	P (kg ha^{-1})	K (kg ha^{-1})	Fertilization	Harvest
2019	5, 7	132	8	101	13 May; 5 Jul	24 Jun; 20 Aug
2019	6	189	14	152	13 May; 5 Jul; 15 Oct	24 Jun; 20 Aug
2020	1–7	146	8	84	29 May; 30 Jun	17 Jun; 14 Aug
2021	1–7	136	7	78	28 May; 7 Jul	22 Jun; 16 Aug

snow gradient method following Lind et al., (2020) (Fig. S3). Gas samples of 20 ml were collected at 10 cm intervals from the snow surface to 2 cm above the soil, using a metal probe (length 61 or 107 cm, ⌀ 2 mm) connected to a 60 ml plastic syringe and injected into pre-evacuated 12 ml glass vials. Measurements were excluded if snow depth was < 10 cm or wind > 2 m/s. Snow porosity was estimated by taking cylindrical samples (⌀ 106 or 112 mm) from two locations per block. Snow depth was measured at the gas and snow sampling locations.

From the gas samples the CO_2, CH_4, and N_2O concentrations were analyzed using a gas chromatograph (HP 7890 series, GC system, Agilent, USA) equipped with flame ionization (FID), electron capture (ECD) and a nickel catalyst detector. Standard gases were used: 3010 ppm for CO_2, 2.01 ppm for CH_4, and 0.99 ppm for N_2O. For N_2O, a small proportion of the chamber measurements (4%) were above the highest standard, but these were not discarded in order to avoid underestimating peak fluxes.

Fluxes from the chamber measurements were calculated based on the ideal gas law and linear regression, and fluxes from the snow-gradient measurements were calculated using Fick's law of diffusion, as detailed in Supplementary Eq. S1-S2. After visual inspection of the data, fluxes were accepted when the gas concentration change with depth was considered linear ($R^2 \geq 0.85$ for chambers, $R^2 \geq 0.75$ for snow gradient). Low fluxes ($\pm$ 0.04 mg CH_4 m^{-2} h^{-1}, $\pm$ 0.04 mg N_2O m^{-2} h^{-1}, <28 mg CO_2 m^{-2} h^{-1}) were accepted regardless of their R^2 value. Approximately 2% of chamber measurements and 11% of snow-gradient measurements were discarded.

3.2. Gross Primary Production and Net Ecosystem Exchange

Net ecosystem exchange (NEE) was measured with a transparent polycarbonate chamber and a Licor LI-850 (Fig. S3, Li-Cor, Inc., Lincoln, NE, USA) gas analyzer, and the data were processed following Trémeau et al. (2024). Air temperature and photosynthetically active radiation (PAR) were recorded inside the chamber and either a 30 cm or 80 cm high chamber was used depending on the grass height. A minimum of four but typically five different light intensities were measured with shrouding that blocked ~25%, 50%, 75% and 100% of the PAR inside the chamber. The last measurement with zero PAR therefore equals R_e measured also with the dark chamber described above but was only used here for deriving gross primary productivity (GPP). At each light intensity, the measuring time was 2 min, and the chamber was vented between every measurement. NEE and R_e fluxes were calculated from linear regression during the closure period and fitting was done using least squares minimization. The measurement was discarded if the normalized root mean square error (NRMSE) was > 0.15 or standard deviation of PAR > 150 μmol m^{-2} s^{-1}. However, NRMSE was not checked if the slope was < 0.1 ppm/s because the fit is typically very noisy with small fluxes. In the results, negative fluxes indicate CO_2 uptake and positive fluxes CO_2 emissions to the atmosphere.

GPP at each light intensity was calculated as the difference between NEE and total ecosystem respiration measured without PAR. To compare the different measurement days, a light response curve was fitted to GPP values measured during one day using a rectangular hyperbolic model:

$$GPP \quad (PAR) = \frac{\alpha * GP\max * PAR}{\alpha * PAR + GP\max} \tag{3}$$

where α (mg CO_2 μmol^{-1} per photon) is the initial slope of the photosynthetic light response and *GPmax* (mg CO_2 m^{-2} s^{-1}) is the theoretical maximum rate of photosynthesis at infinite PAR. The fitting was done with non-linear least squares minimization. Photosynthetic capacity (GP_{1200}) values were calculated with estimated α and GPmax for each measurement day with PAR = 1200 μmol m^{-2} s^{-1}. Flux calculations and the light response curve fitting were done in Python 3.7.7. The scripts are available from Vekuri (2024).

3.3. Supporting environmental measurements

Concurrently with the flux measurements, soil temperature at −5 cm and −10 cm depth was measured manually with a Tenmars TM-80N (Tenmars Electronics Co., Ltd., Taipei, Taiwan), soil moisture (volumetric water content, VWC) at −6 cm depth was measured using an HH2 equipped with a ThetaProbe ML2x (Delta-T Devices Ltd., Cambridge, UK), and plant height was measured adjacent to the flux measurement point. On blocks 1–6, two perforated groundwater pipes (⌀ 50 mm) were installed at 2 m depth, and the water table depth (WTD) was monitored with Solinst Levelogger sensors (Solinst, Ontario, Canada), recording values at 15-minute intervals. The daily average of the two values was used to indicate the WTD of the block, and the measurements from block 5 were applied to block 7, since it had no WTD monitoring.

Leaf area index (LAI) was measured around the GHG collars using a SunScan probe v1.02 R (Delta-T Devices Ltd., Cambridge, UK) in 2019 and 2021, and inside the GHG collars using an LAI-2200C Plant Canopy Analyzer (LI-COR Biosciences, Lincoln, Nebraska, USA) in 2020. Grass yield was determined by harvesting the biomass inside the GHG collars at each harvest (later referred to as collar yield). Additionally, four approximately 1.5 m × 10 m samples were taken from each block outside the GHG collars using a Haldrup experimental plot harvester to determine the average yield for blocks 1–7. Both sample types were oven-dried at 60 °C for 48 h to determine the dry yields.

Soil ammonium (NH_4^+) and nitrate (NO_3^-) content were analyzed from pooled soil samples taken from 0 to 10 cm and 10–20 cm depth from each block on average every three weeks from May till October or November. The soil samples were homogenized in the laboratory and soil extractions were conducted with milli-Q water and 1 M potassium chloride (KCl). The extracts were further analyzed with spectrophotometer (1420 VICTOR3™, PerkinElmer, USA) for NO_3^- and NH_4^+, respectively. The analysis methods are described in Gerin et al. (2023).

Peat depth was obtained as an average from three manual measurements done around each GHG-collar in 2019 and 2020, and the average peat depth for each block was determined from a ground penetrating radar (GPR) survey conducted in July 2020 by Yli-Halla et al. (2022).

4. Statistical analysis

All analyses were conducted using R 4.3.3 (R Development Core Team, 2023). Relationships between GHG fluxes and environmental variables were first explored using Spearman's rank correlations, with separate analyses conducted for nitrate and ammonium concentrations paired with the closest corresponding flux measurement campaign. Correlations were also calculated for peat depth categories: mull soil (<30 cm), shallow peat (30–60 cm), and deep peat (>60 cm), to examine depth-dependent effects.

Linear mixed-effects models (LMMs) were then used to evaluate the influence of peat depth relative to other key environmental variables on GHG fluxes, plant growth, and soil moisture during the snow-free seasons of 2019–2021. Fixed effects included peat depth, soil temperature (-5 cm), soil moisture, water table depth (WTD), plant height, and relevant interactions. This design allowed assessment of both main effects and how peat depth affects these relationships. Due to the limited sample size, LAI and yield were modeled with peat depth as the only fixed effect. Plot nested within block was included as a random effect to account for the spatial structure and the repeated measures, whereas temporal autocorrelation was addressed using a continuous-time first-order autoregressive correlation structure (Pinheiro and Bates, 2000). Spatial autocorrelation was assessed using Moran's I test and variograms, and temporal patterns were checked with autocorrelation function (ACF) plots. Model assumptions were evaluated visually inspecting histograms and Q–Q plots, and variables were Box–Cox transformed when necessary. Multicollinearity among fixed effects was assessed with variance inflation factors (VIF), and variables were mean-centered and scaled when interactions introduced collinearity.

We used backwards elimination to select the best model structure using the Bayesian information criterion (BIC; $\Delta BIC > 2$) for non-nested models and the likelihood ratio test (LRT; $p < 0.05$) for nested models. From the full model, first interactions, then polynomial terms and last full variables were removed until the best model was achieved. Residuals for normal distribution and homogeneity of variances were checked from the models. Maximum likelihood (ML) was used for model selection and the results were reported using restricted maximum likelihood (REML). We report *t*-tests with 95% confidence intervals for all model coefficients, including polynomial terms (Supplementary Tables S2–S4). The overall contribution of each fixed effect was assessed using LRTs with χ^2 statistics and p-values. Marginal R^2 (fixed effects only) and conditional R^2 (fixed and random effects) are reported for each model.

5. Results

5.1. Soil moisture, ammonium and nitrate conditions

Topsoil (0–6 cm) moisture ranged from 0.06 to 0.77 m^3/m^3 during the study period (May-September, 2019–2021), with highest values observed in early summer and autumn and lowest in June–July (Fig. 3D). In 2019 and 2021, soil moisture was on average 12.5% higher in shallow peat (30–60 cm) and 6% higher in deep peat (>60 cm) compared to mull soil, whereas in 2020 moisture was on average 20% higher in both shallow and deep peat compared to the mull soil (<30 cm) (Fig. 3D; Table S5). Heavy rainfall events in summer 2020 caused a peak in the WTD and soil moisture also during mid-summer, with WTD reaching −10 cm at maximum. Throughout the study period thick peat plots maintained shallower and more stable WTD, whereas on plots with shallow peat layer WTD fluctuated more rapidly and dropped much lower during the dry periods (Fig. 3E).

Spearman's rank correlations showed a strong positive relationship between soil moisture and WTD (Fig. S4), with the relationship becoming slightly stronger from mull soil ($\rho = 0.62$) to shallow peat ($\rho = 0.65$) and deep peat ($\rho = 0.73$). Soil temperature was negatively correlated with moisture, and this effect also increased slightly from mull soil ($\rho = -0.47$) to shallow peat ($\rho = -0.52$) and deep peat ($\rho = -0.62$).

Consistent with these correlations, linear mixed-effects models showed that most of the variation in moisture was explained by WTD ($R^2m = 0.42$) and soil temperature ($R^2m = 0.15$), with moisture increasing with higher WTD and decreasing with higher soil

Table 2
Parameters and statistics of linear mixed-effects models for soil moisture at −6 cm (SM), leaf area index (LAI), yield, photosynthetic potential (GP_{1200}), ecosystem respiration (R_e), nitrous oxide (N_2O), and methane flux (CH_4). Effects of predictors—soil temperature at 5 cm (ST), plant height (PH, cm), water table depth (WTD), soil moisture (SM), peat depth (PD), and relevant interactions—were tested. Reported statistics include likelihood ratio χ^2, p-value, and marginal and conditional R^2 (R^2m / R^2c). Values with $p < 0.05$ are considered statistically significant.

Response	Predictor	Effect type	χ^2	p-value	R^2m / R^2c
SM	**Full model**	-	-	-	**0.50 / 0.57**
	ST	Quadratic	158.3	< 0.001	0.15 / 0.17
	PH	Cubic	18.9	< 0.001	0.01 / 0.01
	WTD	Cubic	384.3	< 0.001	0.42 / 0.48
LAI	**Full model**	-	-	-	0.03 / 0.03
	PD	Linear	8.3	0.04	0.03 / 0.03
GP_{1200}	**Full model**	-	-	-	**0.54 / 0.60**
	PH	Cubic	33,78	< 0.001	0.54 / 0.60
R_e	**Full model**	-	-	-	**0.64 / 0.76**
	ST	Cubic	638.9	< 0.001	0.50 / 0.53
	SM	Quadratic	94.8	< 0.001	0.20 / 0.23
	PH	Quadratic	363.0	< 0.001	0.27 / 0.28
	WTD	Cubic	47.1	< 0.001	0.11 / 0.11
	SM × ST	-	11.5	< 0.001	0.04 / 0.04
	SM × WTD	-	42.3	< 0.001	0.02 / 0.02
N_2O	**Full model**	-	-	-	**0.16 / 0.46**
	ST	Cubic	550.5	< 0.001	0.01 / 0.27
	SM	Cubic	41.3	< 0.001	0.04 / 0.28
	WTD	Quadratic	34.4	< 0.001	0.06 / 0.30
	PH	Cubic	73.0	< 0.001	0.01 / 0.28
CH_4	**Full model**	-	-	-	**0.13 / 0.18**
	SM	Linear	158.56	< 0.001	0.13 / 0.18

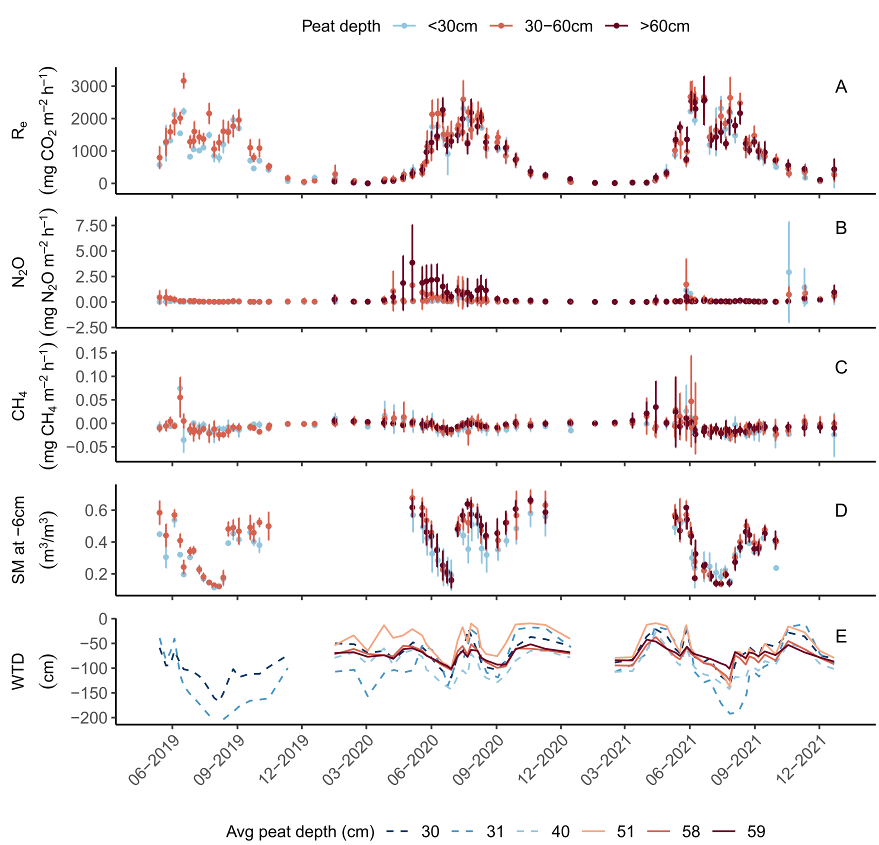

Fig. 3. Average measured ecosystem respiration (R_e) (a), N_2O (b) and CH4 (c) fluxes (mg m^{-2} h^{-1}) and soil moisture (SM, m^3m^{-3}) at −6 cm (d) ± standard deviation in < 30 cm (n = 3), 30–60 cm (n = 17) and > 60 cm (n = 8) peat depth. Daily average water table depth (WTD, cm) in blocks 1–7 and average peat depth per block (e). Grey, solid lines represent fertilizations, dashed line harvests and dark grey line glyphosate application in Sep 2021.

temperature (Table 2, Fig. S5). Plant height had a significant but minor negative effect, mainly between bare soil and 20 cm vegetation. Peat depth or its interactions had no significant effect on moisture.

Nitrate concentrations ranged between 0 and 209 mg N kg^{-1} DW in the 0–10 cm depth and between 0 and 178 mg N kg^{-1} DW in the 10–20 cm depth during the study period. Ammonium concentrations varied between 0–105 mg N kg/DW at 0–10 cm soil depth and 0–53 mg N kg/DW at 10–20 cm soil depth. There were a few episodes of elevated nitrate and ammonium concentrations, mainly in the blocks with the deepest peat (Fig. 4). Nitrate concentrations at both depths were positively correlated with block average peat depth ($\rho = 0.33$ for 0–10 cm, $\rho = 0.61$ for 10–20 cm; Fig S6). In contrast, ammonium concentrations showed only weak negative correlation with peat depth at 10–20 cm depth ($\rho = -0.13$) and no correlation at 0–10 cm depth.

5.2. Peat depth effect on plant growth

In 2019 and 2021, LAI was comparable at all measuring points, but in 2020, plots with deep peat had on average, 17% lower LAI than plots with shallow peat and mull soil (Fig. 5). GP_{1200} showed seasonal variation comparable to LAI. In July 2020, GP_{1200} was also lower on deep peat compared to shallow peat (Fig. 5).

In the LMMs, LAI had a significant negative relationship with peat depth, but peat depth explained only 3% of the variation, (Table 2; Fig. S7). Peat depth did not significantly affect GP_{1200}. Plant height explained 54% of the variation in GP_{1200}, with photosynthetic capacity increasing until plant height reached about 60 cm (Table 2; Fig. S8).

Similarly, peat depth had no significant effect on annual harvest yields in the collars (Table S3). In 2020, yields were highly variable, particularly in blocks 1–4, due to weak growth of young grass. In 2021, yields were more consistent across blocks, though slightly lower than the block means determined from larger sample sizes (Fig. 6).

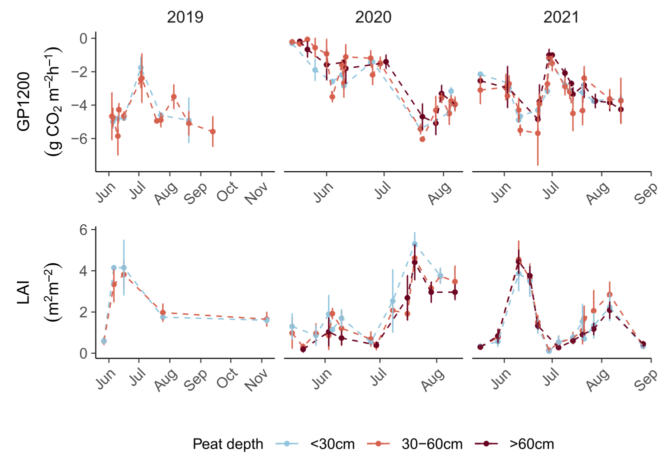

Fig. 5. Average photosynthetic potential (GP_{1200}, mg CO^2 m^{-2} h^{-1}) and leaf area index (LAI, m^{2}m^{-2}) ± standard deviation in < 30 cm (n = 3), 30–60 cm (n = 17) and > 60 cm (n = 8) peat depths. The lower is the value of GP_{1200}, the higher the photosynthetic capacity.

5.3. Peat depth effect on GHG emissions

5.3.1. Ecosystem respiration

R_e was on average 23% and 11% lower on mull soil compared to shallow peat during snow free seasons in 2019 and 2021 respectively, whereas in 2020 R_e was quite comparable at all measuring points (Table S6). On deep peat, R_e was on average 6% lower compared to shallow peat during the snow free seasons in 2020 and 2021.

Spearman's rank correlations showed negative relationship between R_e and WTD in shallow and deep peat ($\rho = -0.2$), whereas no significant

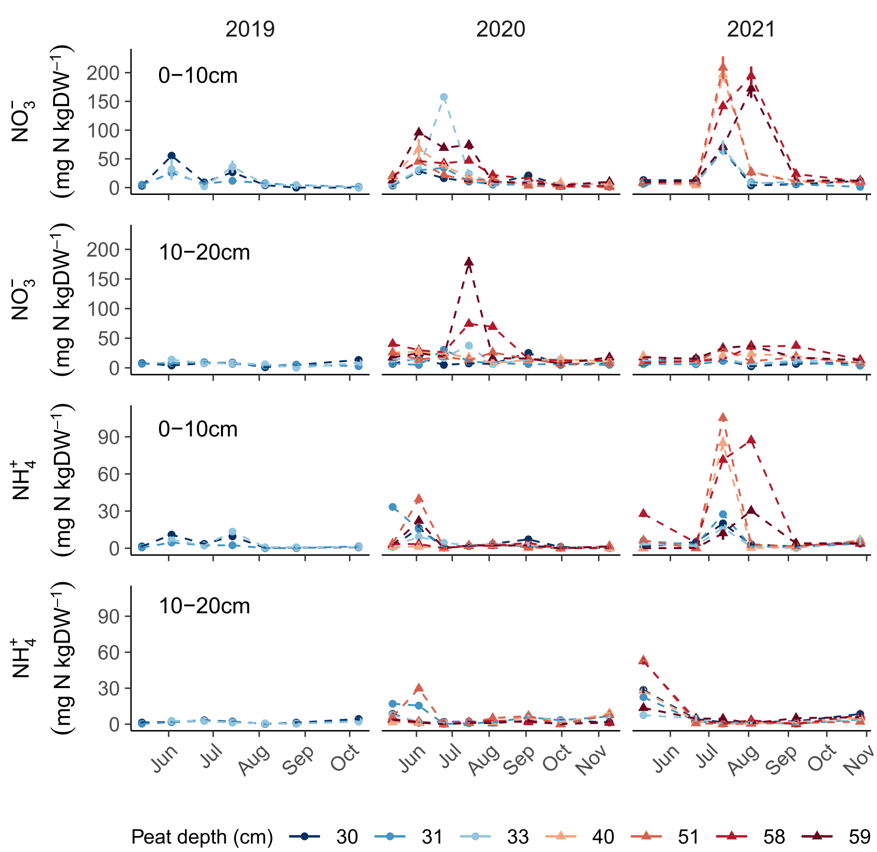

Fig. 4. Average nitrate (mg NO_3-N kg DW^{-1}) and ammonium (mg NH_4-N kg DW^{-1}) concentrations ± standard deviation (n = 3) from 0 to 10 cm and 10–20 cm soil depth, and average peat depth (cm) on blocks 1–7.

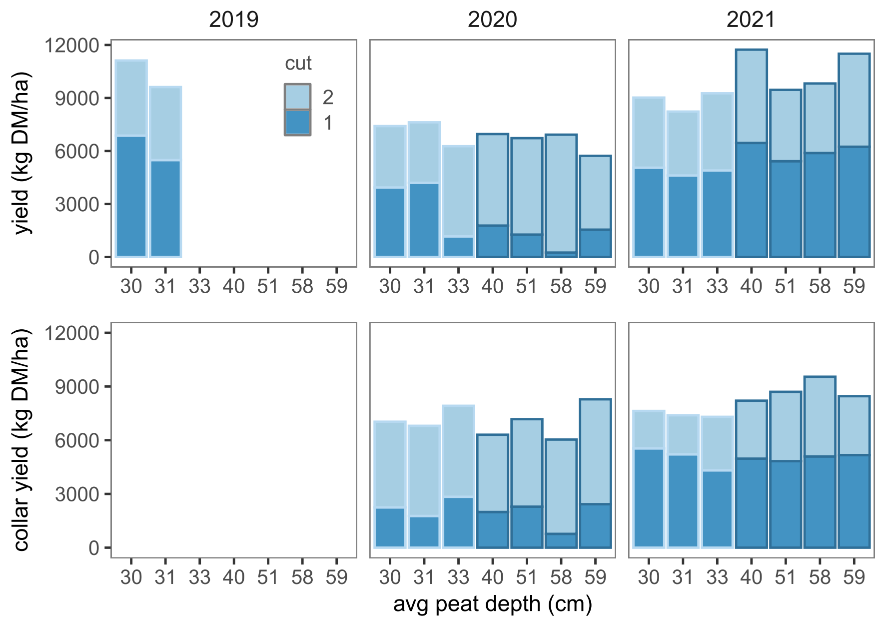

Fig. 6. Average yield (kg DM ha^{-1}) (n = 4) on blocks 1–7 from the block samples outside GHG-collars and from the GHG-collars (collar yield) in first (dark blue) and second (light blue) harvest and average peat depth (cm) on the block. Blocks 1–4 with young grass have dark blue borders and blocks 5–6 with mature grass have light blue borders. In 2019 yield was not determined from GHG-collars or block 7.

correlation was observed in mull soil (Fig. S3). The correlation with soil temperature varied slightly with peat depth, being strongest in mull soil ($\rho = 0.74$) and slightly lower in shallow and deep peat ($\rho = 0.69$ and 0.63, respectively). Also, both nitrate and ammonium concentrations in 0–10 cm depth showed a positive correlation with R_e ($\rho = 0.32$ and 0.31, respectively).

In the LMMs soil temperature and plant height explained most of the variation in R_e (R^2m = 50% and 27%, respectively), with respiration increasing with higher temperature and taller vegetation (Table 2). R_e also showed a significant quadratic relationship with volumetric soil moisture, peaking at approximately 0.4 m^3/m^3 (R^2m = 20%), and decreased with shallower WTD mainly from −0.5 cm from the soil surface (Fig. S9). Interactions among soil moisture, soil temperature, and WTD were observed, while peat depth and its interactions had no significant effect on R_e.

5.3.2. *Methane*

Throughout the study period, CH_4 fluxes were generally small, with the field alternating between being a small source or sink of CH_4 (Fig. 3C). Spearman's rank correlations showed that WTD was positively associated with CH_4 flux (Fig. S3), with the relationship strengthening with peat depth ($\rho = 0.21$, 0.28, and 0.44 for <30 cm, 30–60 cm, and >60 cm, respectively). A similar but less pronounced pattern was observed for soil moisture ($\rho = 0.41$, 0.45, and 0.49). However, in the LMMs, only soil moisture had a significant positive effect on CH_4 flux, explaining 13% of the variation, while WTD, peat depth or its interactions had no significant influence (Table 2, Fig. S10).

5.3.3. *Nitrous oxide*

N_2O fluxes varied both spatially and temporally, with emission peaks observed in 2020 after the first and second fertilization events, and in 2021 after the first fertilization and approximately three weeks after glyphosate application (Fig. 3B). In 2019, fluxes were on average 405% higher in shallow peat than in mull soil during the snow-free season but remained small throughout the year (–0.001–2.04 mg m^{-2} h^{-1}; Table S3). Differences in N_2O fluxes among peat depths were most pronounced in 2020, with fluxes on average 228% higher in shallow peat and 997% higher in deep peat compared to mull soil (Fig. 3B, Table S6). In contrast, 2021 fluxes were generally lowest in deep peat, due to higher peaks observed in mull soil and shallow peat at the end of May and late September (Fig. 3C).

Spearman's rank correlations indicated no consistent trends across peat depths. N_2O fluxes were positively correlated with WTD and soil moisture in shallow peat and showed positive relationships with soil temperature in both mull soil and shallow peat (Fig. S3). Nitrate concentrations at 0–10 cm and 10–20 cm depths were also positively correlated with N_2O fluxes ($\rho = 0.40$ and 0.31, respectively; Fig. S5).

In the LMMs, soil moisture had a significant positive effect on N_2O fluxes up to ~60% VWC but a negative effect at higher moisture levels, while water table depth (WTD) had a positive effect, explaining 4% and 6% of the variation, respectively (Table 2, Fig. S11). Soil temperature and plant height also had significant effects, with N_2O fluxes increasing with temperature and decreasing with plant height between 0–20 cm and 75–100 cm, though they accounted for only ~1% of the variation. Peat depth had no significant effect on N_2O fluxes.

6. Discussion

To accurately estimate national GHG emissions from drained peatlands, it is essential to understand and quantify the GHG exchange dynamics in different types of peatlands. This knowledge is also crucial for life-cycle analyses of food products and for developing climate models and targeting possible emission reductions. Here, we studied in detail the GHG exchange and its driving factors, such as plant growth and soil conditions, at one peat soil site along a gradient of peat depth. We observed some differences in soil hydrology and nutrient conditions along the peat depth gradient, but overall, peat depth did not significantly affect soil moisture, plant growth, or GHG emissions. We will discuss the potential mechanisms and implications of these findings below.

6.1. *A deeper peat layer is associated with higher nitrate concentrations and better water retention*

Our first research question concerned the possible effect of peat depth on plant growing conditions, specifically soil moisture and nutrient status. Although WTD showed clear differences between peat depths, only some periods were observed where topsoil moisture was lower on mull soil compared to shallow or deep peat, and overall peat depth did not have a significant effect on soil moisture. Consistent with this, Pham et al. (2026) reported no influence of peat depth on moisture at a deeper horizon (-30 cm) and found generally good water retention in the organic layer across the field.

Throughout the study period, deeper peat maintained a more stable and higher WTD whereas shallow peat exhibited more variable WTD, with sharp rises during snowmelt and heavy rainfall followed by rapid declines during summer dry periods. Mull soil and shallow peat had higher bulk density and lower porosity in the deeper layers, reflecting more decomposed peat, which is known to reduce the water storage capacity of the soil (Menberu et al., 2018, 2021; Liu et al., 2020). In contrast, thicker peat layers had lower bulk density and greater porosity, which buffer changes in water more effectively (Berglund and Berglund, 2011) and can explain the differences between the peat depths. During the driest periods, WTD dropped well below the peat layer and likely had little effect on the 0–6 cm topsoil, which may explain why the overall effect of peat depth on growing-season moisture was relatively small and non-significant. Despite some variation in SOC and deeper soil conditions among plots, the topsoil generally exhibited similar properties across peat depths, further supporting the finding that peat depth had little influence on surface moisture. Based on the differences in WTD dynamics, moisture conditions are likely to diverge more strongly in deeper soil layers.

There were, however, some periods when mull soils had lower moisture compared with shallow and deep peat and moisture dynamics differed depending on the degree of drying. During the most extreme dry spells (VWC $<$ 40%), topsoil moisture converged across all peat depths. By contrast, under intermediate conditions ($\approx$40–80% VWC), the mull soil often exhibited lower moisture. This supports the idea that mull soil, with its lower storage capacity, reaches water-limited states earlier during drying (Pham et al., 2026). Thus, peat depth appears to influence moisture primarily under moderate drying, whereas under extreme drought all plots converge toward similarly low topsoil moisture.

Soil nitrate concentrations were positively related to peat depth in both 0–10 cm and 10–20 cm layers, with higher peaks occurring mainly between June and August. In the same study site, Yli-Halla et al. (2022) and Pham et al. (2023) reported higher total nitrogen leaching from deep peat blocks compared to shallow ones. Also, Eickenscheidt et al. (2015) measured higher nitrate concentrations from soil with higher C content. Soil micro-organisms use organic carbon as an energy source and can mineralize ammonium from organic materials like manure and crop residues (Girkin and Cooper, 2023). Thus, the higher carbon content in deeper peat can increase the rates of nitrogen mineralization. Also, physical factors like soil temperature, moisture, and aeration are important factors that regulate microbial activity and mineralization (Guntiñas et al., 2012; Girkin and Cooper, 2023). In general, soil moisture correlates positively with nitrogen mineralization until anoxic conditions are reached (Cassman and Munns, 1980). However, we did not identify any clear dependencies between the nitrate concentrations and moisture conditions. Moreover, the higher peaks in deeper peat occurred usually much after fertilization events, suggesting that environmental factors such as soil and weather conditions can regulate the nitrate availability. At the same time, no clear relationship between peat depth and ammonium concentrations was found in either sample depth. Similarly, (Pham et al., 2023) found the differences in loading to be mainly caused by differences in nitrate concentrations, and the ammonium loading to be moderate in this study field compared to other peat and mineral fields. In drained peatlands, increased aeration causes more ammonium to be oxidized to nitrate compared to pristine or restored peatlands, often resulting in higher nitrate than ammonium concentrations (Regina et al., 1999; Daniels et al., 2012).

6.2. Similar plant growth and photosynthesis in shallow and deep peat soil

Our second research question examined whether peat depth affects plant performance in terms of photosynthetic capacity (GP_{1200}), leaf area (LAI) and yield. Although SOC is known to enhance plant growth in mineral soils by improving soil structure, nutrient availability and moisture retention (Loveland and Webb, 2003; Lal, 2006; Pan et al., 2009), the effect of peat depth on plant growth remains poorly studied.

We found only a weak negative relationship between LAI and peat depth and no significant effects on plant height or yield despite the occasional positive effects of deeper peat on moisture and nitrate availability. At our site, SOC concentrations in the top 0–20 cm exceeded 16% across all blocks (Yli-Halla et al., 2022), which is well above the 1–2% thresholds associated with yield limitations in mineral soils (Loveland and Webb, 2003; Oelofse et al., 2015) and can partly explain the limited response. Differences in LAI were only noticeable in 2020, when the young grass in the deep peat grew poorly after the winter and had to be partly resown. This suggests that the differences in LAI were more likely due to the poor establishment of the grass than to differences in the peat depth. Yields in the first harvest were also low especially on the younger grass, whereas plant height was similar across the peat depths throughout the study. The previous winter 2019/2020 had unusually low snow cover and fluctuating temperatures, which likely damaged the grass. The following May and June were also cold and dry, further limiting growth and contributing to the low first-harvest yields.

Consistent with Eickenscheidt et al. (2015), peat depth did not affect photosynthetic capacity (GP_{1200}). Although temperature, soil moisture, and nutrient availability generally regulate photosynthesis (Shurpali et al., 2009, 2010; Heimsch et al., 2021, 2024), neither soil temperature nor moisture significantly influenced GP_{1200} in our model, indicating that photosynthesis was not moisture-limited under the observed conditions during the campaign. This could partly explain why no effect with peat depth was found. Plant height was the main driver of photosynthetic capacity in our model with GP_{1200} increasing until maximal height and decreased after that likely because of reproductive stage. Because of gaps in the LAI data, plant height was used as a proxy, as it is strongly correlated with leaf area index. However, differences in LAI and yields were reflected in the photosynthetic capacity.

Our results provide new field-based insights into peat depth effects on plant growth, complementing previous research that has primarily focused on mineral soils. However, the relatively small dataset, differences in grass age, and year-specific growth suppression limit the detection of subtle peat depth effects. Further long-term research is needed to assess how peat depth and SOC influence yields under varying climatic conditions, particularly to determine whether shallow peatlands are more vulnerable to moisture stress during dry years, a concern that is expected to increase under future climate scenarios.

6.3. Similar GHG emissions across peat depths

Our third research question examined whether peat depth regulates GHG emissions from cultivated peatland. Despite observed differences in soil hydrology and nutrient status, peat depth did not significantly affect ecosystem respiration (R_e). R_e was somewhat lower on mull soil than in shallow peat in 2019, but in 2020 and 2021, R_e was quite similar across peat depths. This is consistent with earlier studies showing that SOC stocks do not directly control emission rates (Leiber-Sauheitl et al., 2014; Eickenscheidt et al., 2015; Yli-Halla et al., 2022; Purviņa et al., 2024).

Deeper organic layers contain less readily decomposable carbon and a higher proportion of recalcitrant humic compounds, which limit microbial activity and CO_2 production despite larger carbon stocks (Nadelhoffer et al., 1991; Bridgham and Richardson, 1992; Waddington et al., 2001). Thus, even though deeper peat layers are exposed to oxygen, decomposition rates remain low due to poorer substrate quality. At our site the topsoil conditions were also relatively uniform, and the C:N ratios were narrow across all blocks, indicating a carbon pool that is easily mineralizable. Higher nitrate peaks were observed in deeper peat and correlated positively with respiration, but regular fertilization likely masked potential depth effects. Similar findings have been reported by Leiber-Sauheitl et al. (2014), who found that neither effective C stocks nor topsoil SOC significantly influenced CO_2 emissions, and by Eickenscheidt et al. (2015), who observed comparable mineralizable C stocks across sites with contrasting SOC levels. Together these factors likely explain why differences in peat thickness did not translate into differences in R_e.

Peat depth can, however, indirectly influence respiration rates through its effects on WTD and soil moisture. Consistent with earlier findings, soil temperature and plant height were the primary drivers of R_e (Kirschbaum, 1995; Silvola et al., 1996; Mäkelä et al., 2022), reflecting the contribution of plant-derived autotrophic respiration, which is related to photosynthetic capacity and plant biomass (Flanagan and Johnson, 2005; Zhang et al., 2013). Given the similarity in plant growth between peat depths, the observed differences in R_e are therefore likely to be mainly driven by differences in heterotrophic respiration from the decomposition of organic matter. WTD had a significant negative effect on respiration, mainly within the upper 50 cm of soil, consistent with previous findings (Weideveld et al., 2021). Correlations indicated that the relationship was stronger in plots with deeper peat, where WTD remained shallower, whereas in mull soil WTD was more frequently too deep to exert an effect. However, the LMM did not show a significant interaction between peat depth and WTD, indicating that higher groundwater levels in deep peat did not systematically constrain R_e. Soil moisture also influenced respiration non-linearly, increasing from 6% to 40% VWC and decreasing between 40% and 80% VWC. During 2019, respiration was lower on mull soil compared to shallow peat, which could partly be due to the drier conditions and moisture limitation. However, in 2020, respiration was similar across peat depths despite the lower moisture on mull soil, highlighting that the effect of moisture can vary considerably between years.

When all years were analyzed together, peat depth did not significantly affect N_2O emissions. This aligns with previous studies comparing shallow and deep peat soils (Yli-Halla et al., 2022; Purviņa et al., 2024), but contrasts with studies suggesting that higher SOC stocks enhance N_2O emissions (Li et al., 2005, 2018; Ye et al., 2016; Kelley et al., 2024). In our study, soil moisture and WTD were the main factors affecting N_2O fluxes, with emissions increasing slightly with moisture up to ~60% and declining rapidly beyond that, and increasing with WTD. Correlations also indicated a positive relationship between N_2O fluxes and nitrate concentrations. This is probably due to denitrification being the dominant pathway for N_2O production in cultivated peatlands (Firestone, 1982; Saggar et al., 2013; Mäkelä et al., 2022), where nitrate is reduced to N_2O under anaerobic respiration, and where the process is strongly regulated by carbon and nitrogen availability, soil moisture, and water table depth (Maljanen et al., 2010; Gerin et al., 2023; Mäkelä et al., 2022; Kelley et al., 2024; Shurpali et al., 2010). The model, however, explained only a small proportion of the variation, highlighting the challenge of modeling N_2O dynamics, which are often dominated by short-lived peaks and can exhibit lagged responses to environmental drivers (Murphy et al., 2022).

We, however, observed major inter-annual variation in N_2O fluxes across peat depths. In 2020, fluxes from deep peat were markedly higher compared to shallow peat or mull soil, while in 2019 and 2021 differences among peat depths were minor or reversed. Several site-specific factors likely contributed to these inter-annual differences. In 2020, suppressed grass growth on the recently established blocks on deep peat reduced plant nitrogen uptake and increased topsoil nitrate concentrations, while high rainfall in July led to shallower WTD and elevated soil moisture. These together created more favorable conditions for denitrification, especially in the deep peat blocks. Similar responses following failed grassland renewal under elevated WTD have been reported by Offermanns et al. (2023), who observed peaks up to 19.76 mg N_2O m^{-2} h^{-1}, while maximum emission rates in our study were lower (9.96 mg N_2O m^{-2} h^{-1}) but still notably high.

In contrast, 2019 and 2021 were characterized by a deeper water table and drier topsoil conditions, and overall lower N_2O emissions. Although deeper peat exhibited higher topsoil nitrate concentrations, these did not translate into N_2O peaks, and no peaks were observed following the second fertilizations, likely due to dry conditions limiting the N_2O production. After the first fertilization in 2019, emissions were higher on shallow peat than on mull soil, coinciding with higher soil moisture on shallow peat. However, in 2021 highest peaks were measured only in mull soil and shallow peat, and after the glyphosate application no major peaks were observed from plots with thicker peat layer. Given the high spatial and temporal variability of N_2O fluxes, and the inherent uncertainty of chamber measurements (Murphy et al., 2022), it should also be noted that it is possible that some short-lived peaks, especially following glyphosate application, may have been missed due to measurement gaps of two to three weeks. Overall, our results indicate that variation in N_2O emissions is mainly driven by site-specific management and weather conditions, rather than peat depth alone. However, peat depth can have indirect effects on the emissions by maintaining higher soil moisture together with higher topsoil nitrate concentrations.

Emissions of CH_4 were low throughout the study period, and the field alternated between a small source and sink. This is typical for cultivated peatlands, where intensive drainage increases soil aeration, suppressing methanogenesis and enhancing CH_4 oxidation by methanotrophs (Maljanen et al., 2007, 2010; Le Mer and Roger, 2001). In line with Yli-Halla et al. (2022) and Purviņa et al. (2024), peat depth had no significant effect on CH_4 fluxes. Instead, variation was primarily explained by soil moisture, with higher water content favoring methanogenesis (Le Mer and Roger, 2001). However, the correlations suggested a somewhat stronger relationship between WTD and CH_4 flux in deeper peat. This indicates that shallower WTD may enhance emissions there, but the interaction was not statistically significant. During most of the study period, the water table remained relatively deep and often below the peat layer and thus limiting its direct influence on surface CH_4 fluxes. Under these conditions, soil moisture represents a more proximal control over methane dynamics, which could explain why WTD and its interaction with peat depth became insignificant in the multivariate model. In general, differences between peat depths were small, and methane contributed minimally to total GHG emissions.

By combining multi-year flux measurements with detailed hydrological, soil and vegetation data, this study provides a comprehensive assessment of peat depth effects on GHG dynamics. Overall, our results indicate that peat depth does not directly regulate GHG fluxes in cultivated peatlands. Instead, the main differences between shallow and deep peat soils were related to hydrology, with deeper peat retaining more water and exhibiting more stable water table dynamics, which at times led to slight decreases in R_e and increases in CH_4 and N_2O emissions. These interactions, however, were not strong enough to translate into systematic differences in emission rates across peat depths. From a management perspective, soils with thicker peat layers can be rewetted more easily, and in the long run, larger carbon pools can be affected. While wetter conditions can promote methanogenesis, the observed increases in CH_4 emissions were small, and the potential reductions in CO_2 emissions from rewetting are likely to outweigh these effects. However, the effects on N_2O emissions should be further studied.

7. Conclusions

Over three years of monitoring soil conditions, plant growth, and GHG fluxes in a northern drained agricultural peatland with varying peat depth, we found that hydrology and management practices, rather than peat depth, were the main drivers of emissions. While deeper peat had higher water holding capacity, more stable water table dynamics, and higher topsoil nitrate peaks, these differences did not translate into systematic effects on soil moisture, plant growth, or overall GHG emissions. Notably, poor grass establishment on deeper peat plots in 2020 likely contributed to elevated N_2O fluxes that year. Our results suggest that categorizing cultivated peatlands by peat depth is unlikely to improve the accuracy of national GHG inventories, as shallow peat soils can produce emissions comparable to deeper ones. Instead, our findings indicate that hydrological parameters of cultivated peatlands should be considered in inventory estimates. However, this conclusion primarily applies to grasslands; full crop rotations and the effects of peat depth during crop establishment may differ and warrant further study. Also, future studies would benefit from dynamic modelling to show how GHG exchange will change in the future as the peat gradually decreases.

CRediT authorship contribution statement

Miika Läpikivi: Writing – review & editing, Visualization, Data curation. **Erkki Joki-Tokola:** Writing – review & editing, Funding acquisition. **Henriikka Vekuri:** Writing – review & editing, Formal analysis. **Jari Liski:** Writing – review & editing, Funding acquisition. **Annalea Lohila:** Writing – review & editing, Funding acquisition. **Hannu Marttila:** Writing – review & editing, Funding acquisition. **Timo Lötjönen:** Writing – review & editing. **Liisa Kulmala:** Writing – review & editing, Supervision, Funding acquisition, Conceptualization. **Milla Niiranen:** Writing – original draft, Visualization, Formal analysis, Data curation, Conceptualization. **Maarit Liimatainen:** Writing – review & editing, Supervision, Project administration, Funding acquisition, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.agee.2026.110400.

Data availability

Data will be made available on request.

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