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Modeling of greenhouse gas emissions from paludiculture in rewetting peatlands is improved by high frequency water table data

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Abstract

Rewetting drained peatlands can reduce CO_2 emissions but prevents traditional agriculture. Crop production under rewetted conditions may continue with flood-tolerant crops in paludiculture, but its effects on greenhouse gas (GHG) emissions compared to rewetting without further management are largely unknown. This study was conducted between 2021 and 2022 on a fen peatland in central Denmark. At the study site, three harvest/fertilization management treatments were implemented on Reed Canary Grass (RCG) established in 2018. Measurements of CO_2 and CH_4 emissions were conducted biweekly using a transparent manual chamber connected to a gas analyzer and manipulating light intensities with four shrouding levels. Although this was a rather wet peatland (-8 cm mean annual WTD), the site

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was a CO_2 source with a mean net ecosystem C balance (NECB) of 6.5 t C ha^{-1} yr^{-1} across treatments. Model simulation with the use of high temporal resolution water table depth (WTD) data was able to better capture ecosystem respiration (R_{eco}) peaks compared to the use of mean annual WTD, which underestimated R_{eco}. Data on pore water chemistry further improved statistical linear models of CO_2 fluxes using soil temperature (Ts), WTD, ratio vegetation indices and PAR as explanatory variables. Significant differences in CO_2 emissions and water chemistry parameters were found between studied blocks, with higher R_{eco} corresponding to blocks with higher pore water nutrient concentrations. Methane emissions averaged 113 kg of CH_4 ha^{-1} yr^{-1}, equivalent to 11.3% of the total carbon emission in CO_2 equivalents. Because of large heterogeneity among the experimental blocks no significant treatment effect was found, however, the results indicate that biomass harvest reduces GHG emission from productive rewetted peatland areas in comparison with no management, whereas on less productive areas it is beneficial to leave the biomass unmanaged.

1 Introduction

Peatlands are an essential component of the global C cycle. Covering only 3% of the terrestrial surface they store ~600 Gt of C, equivalent to 30% of the global soil C pool and exceeding the C stored in vegetation by ~150 Gt (Yu et al., 2010; Scharlemann et al., 2014; Erb et al., 2018; Leifeld and Menichetti, 2018). Northern temperate peatlands can be classified as bogs or fens and store 21.9 Gt C (Leifeld and Menichetti, 2018). While bogs are rain fed and nutrient poor, fens receive drain and ground water from the upland and occasionally from the streams under flooding conditions making them minerotrophic with a pH close to neutral because the incoming waters carry minerals released from surrounding soils and sediments. Under high nutrient concentrations, fens are dominated by grasses and sedges such as *Phragmites sp.* and *Cladium sp.* (Page and Baird, 2016; Kreyling et al., 2021).

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Peatland drainage creates aerobic conditions leading to peat mineralization, and consequently soil C is emitted as CO_2 to the atmosphere (Page and Baird, 2016), and dissolved C and N compounds are leached from the soil (Cabezas et al., 2012; Liu et al., 2019). Emissions from drained peatlands are estimated globally to 785 Mt CO_2 equivalents and the water table is considered the main controlling factor (Zhong et al., 2020; Evans et al., 2021) with higher water tables resulting in lower CO_2 emissions (Tiemeyer et al., 2020; Evans et al., 2021; Koch et al., 2023). However, other factors such as soil temperature (Ts), vegetation, and nutrient status may also affect CO_2 emissions from drained peat soils (Wilson et al., 2016; Rigney et al., 2018; Bockermann et al., 2024). When peatlands are drained, the peat bulk density increases (Liu et al., 2019; Loisel and Gallego-Sala, 2022), and peat chemistry changes leading to decreasing C:N ratio, increased concentrations of humic compounds, polyphenols, dissolved organic C (DOC) and N (DON), and NH_4. The changes in peat chemistry may in turn enhance organic matter mineralization (Cabezas et al., 2012; Liu et al., 2019; Zak et al., 2019), and the release of nutrients along with higher bacterial and fungal activity increases CO_2 emissions (AminiTabrizi et al., 2022; Song et al., 2022).

The importance of peatlands for C storage and GHG emission mitigation, as well as other environmental services, has sparked an interest in peatland restoration with focus on rewetting (Page and Baird, 2016; Andersen et al., 2017). While rewetting reduces CO_2 emissions, it also may lead to increased CH_4 emissions (Wilson et al., 2016; Zhong et al., 2020; Darusman et al., 2023). The CO_2 / CH_4 emission trade-off depends on the water table, the origin of the water (bog/fen), type of vegetation (Rigney et al., 2018; Purre et al., 2019), its nutrient status (Wilson et al., 2016; Tiemeyer et al., 2020), as well as gradual changes in the microbial community following rewetting (Putkinen et al., 2018; Hemes et al., 2019; Emsens et al., 2020; Urbanova and Barta, 2020).

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Rewetting can be achieved through different pathways depending on the land use in the peatland after raising the water table. Often, peatlands are rewetted either without altering the established plant community or by attempting to reestablish the plant communities present in pristine peatlands. Paludiculture has been suggested as an alternative land use of rewetted peatlands enabling continued agricultural biomass production under low or high management intensity (Tanneberger et al., 2020; Ziegler, 2020). It can also reduce CO_2 emissions due to the water-saturated conditions of the peat soils (Ren et al., 2019; Tanneberger et al., 2020; De Jong et al., 2021) while producing biomass for renewable energy such as biogas production (Dragoni et al., 2017; Ren et al., 2019; Hartung et al., 2020) or material that can be used as a green alternative in the building industry. Paludiculture may also have the potential to remove excess nutrients from rewetted peatlands (Giannini et al., 2017; Vroom et al., 2018; Geurts et al., 2020). Large variation in quantified annual GHG emission from different land use of rewetted peatlands have been reported and further studies are needed to establish emission factors for them (Bianchi et al., 2021).

It is well known that greenhouse gas emissions from rewetted peatlands are influenced by their nutrient content and water table level reflected by IPCC Tier 1 emissions factors (Wilson et al., 2016). Mean annual water table depth has also been used to predict the net ecosystem carbon balance (NECB), but much uncertainty remains unexplained (Tiemeyer et al., 2020; Evans et al., 2021; Koch et al., 2023). The complexity and temporal resolution of gap filling models can also influence the NECB estimates (Karki et al., 2019; Liu et al., 2022) and it is highly uncertain how water table dynamics during the year, as well as nutrient status and different management practices affect annual emission budgets. Consequently, the objectives of this study were to: (1) determine the NECB of reed canary grass (RCG) production under three harvest and fertilization management strategies during the third year after establishment in a peatland with shallow WTD (2) assess model performance in gap

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filling biweekly measurements of ecosystem respiration (R_{eco}) and gross primary productivity (GPP) using high temporal resolution data on water table depth (WTD) and (3) investigate the relation of soil water chemistry with R_{eco} and GPP. We hypothesized that fertilization and harvesting RCG would increase C emissions compared to no RCG management, that the use of high-temporal frequency data on water table depth (WTD) would improve model prediction of ecosystem respiration (R_{eco}), and that the addition of soil pore water chemistry parameters as explanatory variables would improve the explanation of C fluxes.

2 Materials and methods

2.1 Study area

This study was conducted from May 2021 to May 2022 at a riparian fen peatland located in the Nørreå valley, Vejrumbro, Central Jutland, Denmark (56°26´15.3´´N, 9°32´44.1´´E) (Fig 1). The site was drained in the 1930s and used for agriculture predominantly under grassland rotation and grazing. The field became gradually wetter because of land subsidence, and the water level was largely controlled by the Nørreå stream, located at the southern border of the peatland (Malinowski et al., 2015). After 2018, maintenance of the drainage ditches stopped and the mean annual WTD across the experimental plots gradually increased during the following years reaching -8 cm during the study year, with a mean minimum of -35 cm in the summer and a mean maximum of 8 cm in the winter (Fig 2a). The mean air temperature and total precipitation, measured at the Foulumgard meteorological station (Danish Meteorological Institute), located 6 km from the study site, were 9 °C and 709 mm, respectively. The peat layer at the study site has an average depth of 2 m covering up to 10 m of gyttja (Mashadi et al., 2024). The physicochemical characteristics of the peat were measured for the top 1 meter of the soil as part of a previous study (Table 1) by Nielsen et al. (2023b).

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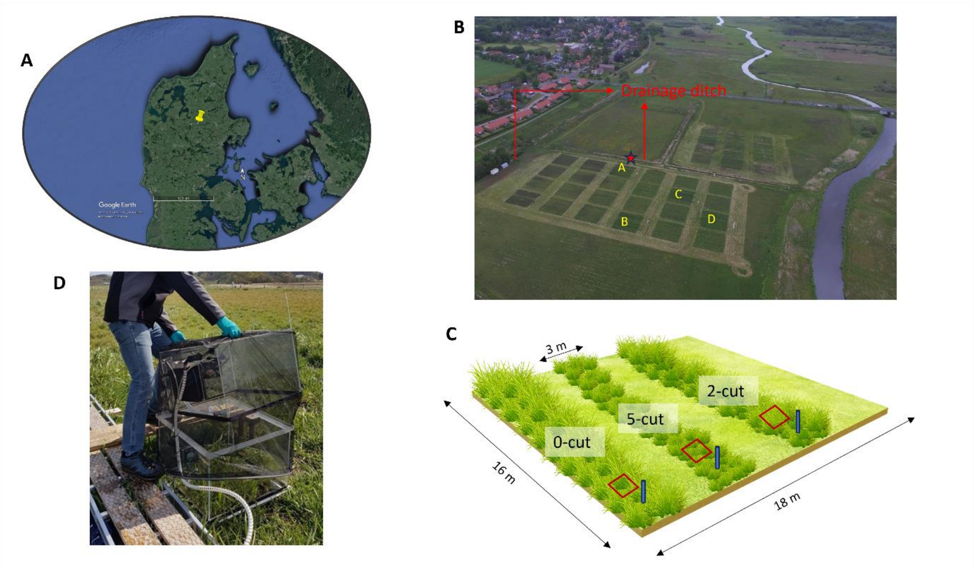

Figure 1. A, map of Denmark indicating the study site location © Google Earth; B, aerial photograph of study site, letters indicate the four studied blocks, and red star indicates where the ditch water samples were taken from; C, diagram of one of the blocks showing the three randomized harvest treatment plots (0-cut, 2-cut, and 5-cut) and the location of collars (red squares) and piezometers (blue cylinders); D, transparent chamber with shroud used for gas measurements.

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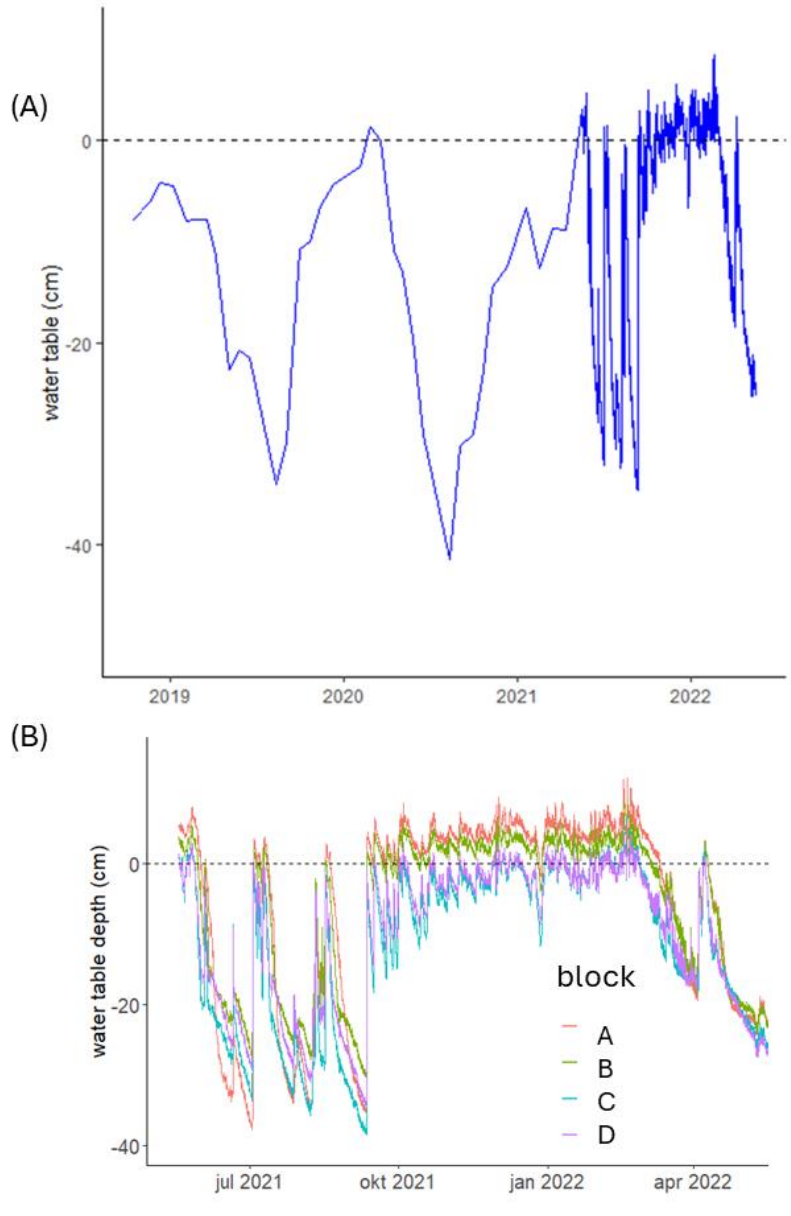

Figure 2. (A) Water table depth at the study site from October 2018, from Oct 2018 to Apr 2021 data was collected biweekly, from May 2021 until May 2022 data was collected hourly. (B) Hourly water table depth per plot during the studied year. Values are average of plots. Colors indicate different blocks.

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Table 1. Soil physicochemical characteristics at the study site of the four studied plots (A-D).

Plot	OM	pH	ρb	TC	TN	C:N
	%		g cm^{-3}	g kg^{-1}	g kg^{-1}	g kg^{-1}
A	85	5.6	0.15	440	26	17
B	83	6.0	0.15	430	28	14
C	70	6.2	0.18	374	24	15
D	75	6.2	0.13	401	27	15
Mean	78	6.0	0.15	411	26	15

†OM, organic matter; ρb, bulk density; TC, total C; TN, total N; C:N, carbon to nitrogen ratio.

2.2 Experimental design

Four blocks (indicated by A, B, C and D on Fig 1B) were established with reed canary grass (RCG, *Phalaris arundinacea,* cultivar Lipaula) in 2018 as part of a larger field experiment. Each block had six randomly placed plots with treatments of different combined effects of harvest and fertilization whereof only three (0-cut, 2-cut, 5-cut) were used for this study. Harvest and fertilization dates can be seen in Figure 3. The harvested plots were fertilized with 200 kg N ha^{-1} and 178 kg K ha^{-1} in total, given as NPK 18-0-16 in equal split doses. Thus, the 2-cut and the 5-cut received 100 kg N ha^{-1} and 40 kg N ha^{-1} for each cut, respectively, while the 0-cut did not receive any fertilizer. The dimensions of the blocks and plots were (16 x 18 m), and (16 x 3 m), respectively (Fig 1C). Further details of the experimental design can be found in Nielsen et al. (2021). At each plot, one 55 x 55 cm collar was installed to 10 cm depth to facilitate closed, none-steady-state chamber measurements of net CO_2 and CH_4 fluxes. A piezometer with a screen from 5 cm to 100 cm soil depth was installed 10-20 cm away from the collar for soil water sampling. Ts at 5 cm soil depth and WTD were continuously measured using Ts dataloggers (HOBO Pendant temperature/light 64K data logger; Onset Corporation, Massachusetts, USA), and Leveloggers (Levelogger 5 Junior; Solinst Canada Ltd, Ontario, Canada), respectively. Perforated gauge tubes for leveloggers sealed with lids and soil temperature loggers were installed in 2020 inside the collars.

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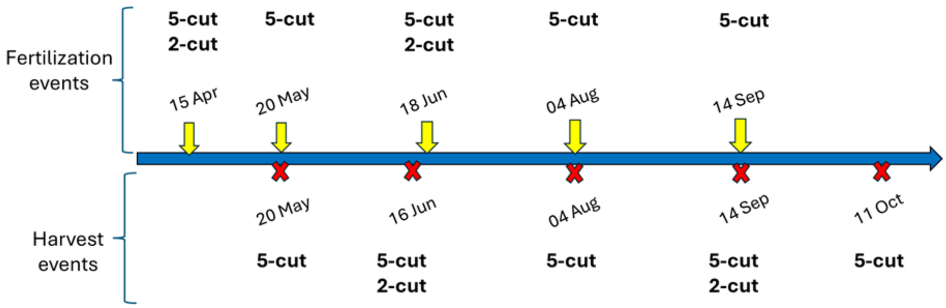

Figure 3. Timeline of fertilization and harvest events applied to the 2-cut and 5-cut harvest treatments during 2021 at the studied blocks.

2.3 Net carbon dioxide and methane flux measurements

The CO_2 and CH_4 measurements were performed biweekly +/- one week depending on meteorological conditions between the 28th of May 2021 and the 14th of June 2022. A total of 26 campaign measurements were undertaken. Fluxes were measured using a fully transparent chamber (60 cm x 60 cm x 41 cm) made of Plexiglass and equipped inside with a photosynthetic active radiation (PAR) sensor (190-SA; Li-Cor Inc., Lincoln, NE, USA), a temperature sensor, and an air mixing fan. Further details of the chamber design and how the temperature was controlled during operation can be found in Elsgaard et al. (2012). The chamber was connected to an LGR-ICOS^{TM} GLA131-GGA microportable gas analyzer (ABB Ltd.), which simultaneously measured water vapor corrected CO_2 and CH_4 (i.e., dry fractions) at 1 Hz resolution. Chamber deployment was 120 sec per measurement. All data were stored in a Campbell CR1000X data logger (Campbell Sci. Logan, UT, USA) with the same timestamp. In order to fit the RCG inside the chamber during growth, a chamber extension with the same dimensions as the measurement chamber was used during all measuring campaigns, i.e. total chamber height with the extension was 82 cm. Measurements were performed between 10:00 am and 3:00 pm on predominantly clear days without precipitation. Measurements were conducted during constant PAR conditions, when possible,

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by timing measurements such that changing cloud conditions were avoided. For each campaign and at each soil collar, fluxes were measured corresponding to four PAR levels by using net shrouds and an opaque cover as described by Kandel et al. (2017). This resulted in four flux measurements, one under fully transparent conditions which corresponded to net ecosystem exchange (NEE), a second under ca. 50% blocked PAR, a third under ca. 75% blocked PAR, and a fourth under 100% blocked PAR equivalent to R_{eco}. Between PAR levels plants were given one minute to adapt to the new PAR conditions while the chamber was lifted on one side, allowing air circulation and bringing CO_2 and CH_4 concentrations to atmospheric levels.

All fluxes were calculated using the Flux package 0.3-0.1 (Jurasinski et al., 2022) in R (R Core Team (2023), R version 4.3.0). Inspection of fluxes revealed that fluxes were mostly linear, and flux rates were therefore calculated based on linear regression. For low CO_2 fluxes (<100 mg CO_2 m^{-2} h^{-1}), fluxes with an $R^2 < 0.6$ and a nrmse > 0.1 were removed, while for high CO_2 fluxes (>100 mg CO_2 m^{-2} h^{-1}), fluxes with an $R^2 < 0.9$ and a nrmse > 0.1 were identified and the PAR and CO_2 flux were manually inspected. If sudden changes in the PAR occurred during the 2 min measurement period or if the flux curve indicated a possible leakage, flux data were discarded. These criteria resulted in 3% of the calculated CO_2 fluxes being removed. In the case of CH_4, all calculated fluxes had R^2 values higher than 0.9, therefore no fluxes were removed based on non-linearity, if a possible leakage was identified fluxes were removed, resulting in 1.6% of the fluxes removed. For further calculations, only the CH_4 fluxes measured under 100% PAR blocked (opaque conditions) were used.

2.4 Biomass measurements

Spectral reflectance was measured in all collars biweekly at gas sampling days and before and after harvest events using a portable crop sensor (RapidSCAN CS-45; Holland Scientific

Inc., Lincoln, NE, USA), which was held 30 cm above the canopy and horizontally rotated 45° while performing measurements to cover all vegetation inside the collar. Approximately 30 scans were taken per collar and their mean values were used to calculate the ratio vegetation index (RVI) as the ratio of near-infrared to red light reflectance. The RVI has been used as a proxy for photosynthetically active biomass and it has been used in photosynthesis and ecosystem respiration models (Kandel et al., 2017; Karki et al., 2019). Hourly RVI values were obtained by linearly interpolating biweekly RVI measurements, hourly values were used in GPP and R_{eco} modelling. Fresh weight yield and dry matter content was determined by harvesting the biomass inside the collars at respective cuts. This biomass was analyzed for total N and C with a Vario Max CN (Elementar Analysesysteme GmbH, Hanau, Germany). Dry matter yields (Table A1) were multiplied by percentage C to obtain the yield in C ha^{-1} yr^{-1} as part of the CO_2-C budget. The sum of yields from individual cuts per treatment was considered as the annual yield.

2.5 Gap filling models and annual budgets

The measured NEE CO_2 fluxes were partitioned into GPP and R_{eco}. The GPP was calculated as NEE – R_{eco} and it was calculated for all PAR levels. From an atmospheric perspective we always consider R_{eco} positive, and GPP negative while NEE can be either positive (ecosystem carbon source) or negative (ecosystem carbon sink). The net ecosystem carbon balance (NECB) was calculated as the sum of the NEE plus the harvested yields of the 2-cut and 5-cut treatments. For calculation of annual budgets, three models from previous studies (one for GPP and two for R_{eco}, see below), which used RVI, Ts, and WTD were selected as explanatory variables. Additionally, a fourth model was developed based on a modification of the two selected R_{eco} models. The GPP was modelled based on Karki et al. (2019) (model 1).

$$GPP = \frac{GPP_{max} * PAR}{k + PAR} * \left(\frac{RVI}{RVI + \alpha}\right) * FT \qquad \text{(model 1)}$$

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Where GPP is in mg CO_2 m^{-2} h^{-1}, *RVI* is the ratio vegetation index, *k* is the PAR value at which GPP reaches 50%, α is a fitted parameter, and *FT* is a linear temperature dependent function set to 0 when temperature < -2 °C and to 1 when temperature > 10 °C (Kandel et al. 2017).

R_{eco} was modelled based on Karki et al. (2019) with RVI and Ts as input variables (model 2), based on Rigney et al. (2018) with WTD and Ts as input variables (model 3), and with our developed model, which included RVI, WTD and Ts as input variables (model 4).

$$Reco = t1 + (a * RVI) * e^{\left[b*\left(\frac{1}{T_{10}-T_0} - \frac{1}{T_S-T_0}\right)\right]} \qquad \text{(model 2)}$$

$$Reco = t1 * e^{\left[b*\left(\frac{1}{T_{10}-T_0} - \frac{1}{T_S-T_0}\right)\right]} + (WTD + \mathrm{c})^2 \qquad \text{(model 3)}$$

$$Reco = t1 + (a * RVI) + [(WTD - WTD_{max}) * \mathrm{c}]^2 * e^{\left[b*\left(\frac{1}{T_{10}-T_0} - \frac{1}{T_S-T_0}\right)\right]} \qquad \text{(model 4)}$$

Where R_{eco} is in mg CO_2 m^{-2} h^{-1}*RVI* is the ratio vegetation index, *WTD* is the water table depth (cm), WTD_{max} is the maximum WTD (cm), *t1*, *a*, *b*, and *c* are fitted parameters, *t1* has a lower limit set at 1, while all other fitted parameters are without upper and lower limits. T_{10} is the reference temperature set to 10 °C, T_0 is the zero-respiration temperature set to -46 °C, and T_s is the soil temperature (°C) at 5 cm depth.

Each R_{eco} model was fitted to data obtained biweekly using non-linear regression (non-least square) in R (R Core Team (2023), R version 4.3.0) for each plot independently. Annual CO_2 budgets were calculated using the parameterized models, hourly Ts, WTD, and RVI. Model performance was evaluated by comparing the measured GPP and R_{eco} with the modelled values using the following indices: Nash-Sutcliffe efficiency, which indicates how well the plot of observed versus simulated data fits the 1:1 line, with more accurate models having values closer to 1, corrected Akaike information criterion (AICc), normalized root mean square error, and R^2 using the hydroGOF package in R (Zambrano-Bigiarini, 2020). Based on these criteria, the best performing R_{eco} model was used to calculate the annual CO_2 budget. In addition, field models of R_{eco} (model 4) and GPP were parameterized by pooling data from all blocks and treatment plots. For CH_4, measured fluxes were linearly interpolated to obtain the annual CH_4 budget.

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We tested the sensitivity of the best performing R_{eco} model (model 4) to the frequency of WTD data either using (a) hourly WTD, Ts, and RVI (b) annual mean WTD with hourly Ts and RVI, and (c) annual mean WTD, annual mean Ts, and hourly RVI.

2.6 Water chemistry

Soil pore water was collected biweekly at the same time as the gas campaigns and analyzed for total organic C (TOC), dissolved organic C (DOC), total nitrogen (TN), total dissolved nitrogen (TDN), nitrate-N, (NO_3), ammonia-N (NH_4), total P (TP), total dissolved P (TDP), Fe, pH, Electroconductivity (EC), and turbidity. Pore-water samples were collected immediately after each GHG measurement from piezometers installed 20 cm from each GHG collar. Water samples were extracted through a tube fitted at the end with an aquarium air stone (Air Stone Economy Cylinder 4 X 5 cm, Aquakoi / JV Trading Aps) placed 20 cm below the water table in each piezometer. An additional sample was collected from a ditch draining the peatland. A total of 13 samples were collected per campaign for a total of 338 samples. Upon collection, part of the sample was filtered using 0.45 µm pore size filter. The unfiltered samples were analyzed for pH and electroconductivity (EC) following the Danish Standards DS287 and DS288, respectively, turbidity, TN following Best (1976), TP using the Danish Standard, DS291 photometric method (Dansk Standard, 2004), TOC using a total organic C analyzer (TOC-VCPH; Shimatzu Corporation, Kyoto, Japan), and Fe by ICP emission spectrometer (iCAP 6000 series; Thermo Fisher Scientific, Inc., Walthman, Massachusetts, USA). The filtered samples were analysed for DOC with a (TOC-VCPH; Shimatzu Corpotation, Kyoto, Japan), TDN and NO_3 (Best, 1976), TDP by the Danish Standard, DS291 photometric method (Dansk Standard, 2004), and NH_4 following Crooke and Simpson (1971).

2.7 Statistical analysis

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Statistics were performed in R (R Core Team (2023), R version 4.3.0). Effects were considered significant if *p value* < 0.05. Normality assumptions were evaluated with Q-Q plots, histograms, and residual plots. Kruskal-Wallis tests were used to test the effect of harvest treatment and block on R_{eco}, GPP, NEE, and NECB. Correlations and principal component analysis (PCA) were used to establish relationships between water chemistry parameters, R_{eco}, GPP, NEE, Ts, RVI, PAR, WTD, and CH_4.

ANOVA and Tukey tests were used to determine differences between water chemistry parameters among blocks and harvest treatments. The effects of each water chemistry parameter on R_{eco} and GPP were tested with linear mixed models. Each water chemistry parameter was added one by one as a fixed factor to the base models shown below as models 5 and 6, and the performance of the model including each water chemistry parameter was compared to the base model. The R_{eco} base model included WTD, Ts, and RVI as fixed factors and the measuring campaign and replicate block as random factor (model 5), while the GPP base model included PAR, Ts, and RVI as fixed factors and measuring campaign and replicate block as random factors (model 6).

(model 5) $Reco = Harvest\ treatment + WTD + Ts + RVI + (campaign) + (R.Plot)$

(model 6) $GPP = Harvest\ treatment + PAR + Ts + RVI + (campaign) + (R.Plot)$

Likelihood ratio tests were used to establish if there was a significant improvement of the base models by adding the water chemistry parameters, if this was the case, the R^2 and root mean square error (RMSE) were calculated. Outliers of the water chemistry data were identified as being larger than 3 times the standard deviation for each parameter independently excluding 1% of the data from the analyses.

3. Results

3.1 Model performance

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Measured R_{eco} was best described by model 4 in 11 out of the 12 studied plots based on the NSE and in 10 out of 12, based on the AICc (table 2). The other calculated indices (R^2, and NRMSE) also supported model 4 as the best overall performing model. When WTD was excluded as seen in model 2 compared to model 4, the overall performance was reduced as indicated by lower NSE for most plots except for plot C 5-cut and plot D 0-cut (Table 2). Model 3, where RVI was excluded, had the lowest performance of the three tested models. In general, the 0-cut plots provided the best model performances with NSE and $R^2 > 0.9$ and the lowest AICc, while the 2-cut and 5-cut plots had lower model performances (between 0.74 and 0.92 NSE). The performance results for the GPP models had R^2 values that ranged between 0.81 and 0.96 (Table A3).These results show that the GPP models had R^2 values that ranged between 0.81 and 0.96.

R_{eco} was positively correlated with Ts and RVI, and negatively correlated to WTD (lower WTD = deeper WTD). On the other hand, GPP was negatively correlated to Ts, RVI, and PAR. These expected relationships seen in PCA plots (Fig 4) and correlations (Fig. A1) support why the variables in models 1-4 were selected and parameterized in this study. The fitted parameter values of the best performing R_{eco} model and the GPP model varied between plots (Fig 5). For the R_{eco} model, the *b* parameter was near its maximum value in most plots, while for the GPP model, the *k* parameter was near its maximum in most plots.

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Table 2. Evaluation of three R_{eco} models parameterized for each plot by four different performance indices.

Block	Treatment	Model 2				Model 3				Model 4			
		R2	NRMSE	NSE	AICc	R2	NRMSE	NSE	AICc	R2	NRMSE	NSE	AICc
A	0	0.95	21.4	0.95	980	0.96	19.4	0.96	1216	0.97	16.7	0.97	943
	2	0.85	39.2	0.84	1090	0.7	54.7	0.7	1424	0.88	34.8	0.88	1073
	5	0.71	53.5	0.71	1247	0.84	39.9	0.84	1481	0.82	42.5	0.82	1211
B	0	0.95	21.6	0.95	1029	0.93	26	0.93	1331	0.98	15.4	0.98	978
	2	0.71	53.3	0.71	1261	0.67	57.4	0.67	1570	0.74	50.7	0.74	1255
	5	0.83	40.7	0.83	1113	0.85	39	0.85	1389	0.85	38.3	0.85	1106
C	0	0.96	19.3	0.96	1109	0.92	27.7	0.92	1446	0.96	18.7	0.96	1106
	2	0.77	47.8	0.77	1175	0.71	53.8	0.71	1565	0.81	43.9	0.8	1163
	5	0.84	39.6	0.84	1227	0.84	39.9	0.84	1519	0.84	39.6	0.84	1229
D	0	0.9	32.1	0.9	1030	0.87	36.4	0.87	1348	0.9	32.1	0.9	1032
	2	0.82	41.8	0.82	1229	0.81	43.3	0.81	1533	0.88	34.3	0.88	1198
	5	0.91	31.1	0.9	1153	0.84	40.1	0.84	1484	0.92	28.5	0.92	1142

A, B, C, and D are the four blocks, The three harvest treatments at each block (plots) are 0, 2, and 5. The four indexes of model evaluation are: R^2, normalized root mean square of error (NRMSE), Nash-Sutcliffe efficiency (NSE), and corrected Akaike information criterion (AICc).

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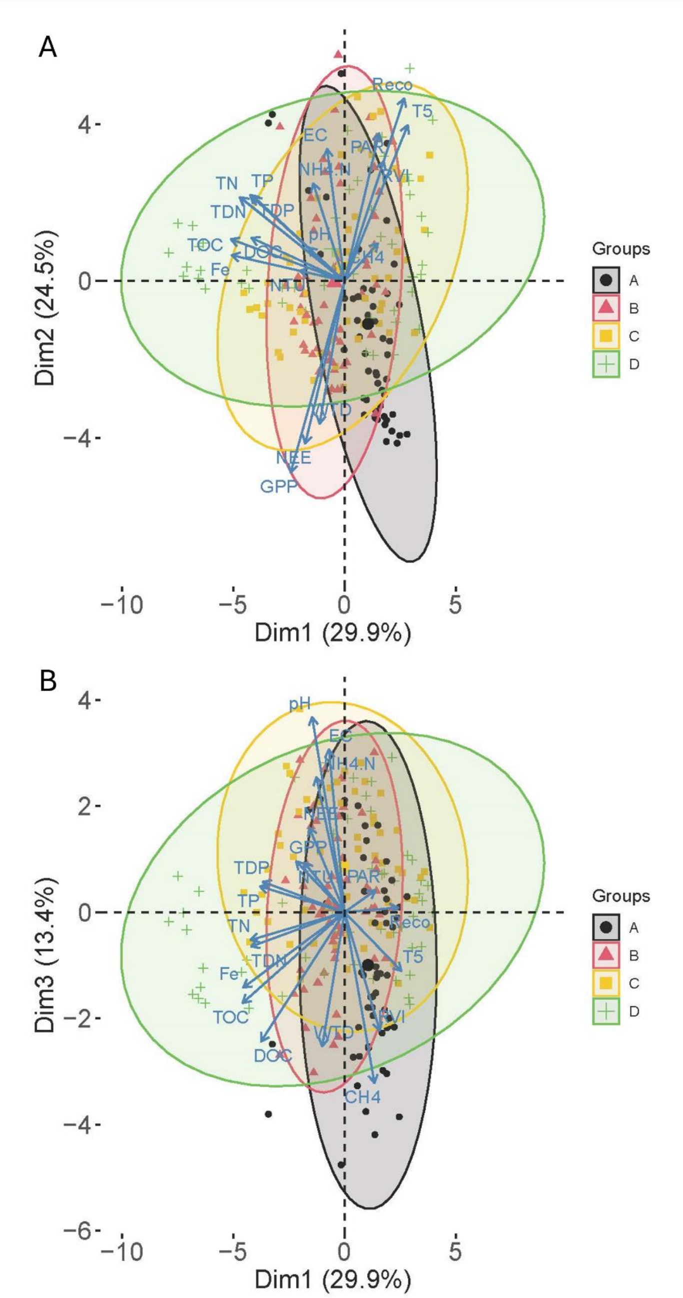

Figure 4. Principal component analysis plots. PC1 vs PC2 (A), PC1 vs PC3 (B). Variability explained by each PCA is the value in parenthesis. Colors represent the four studied blocks. Harvest treatments are combined.

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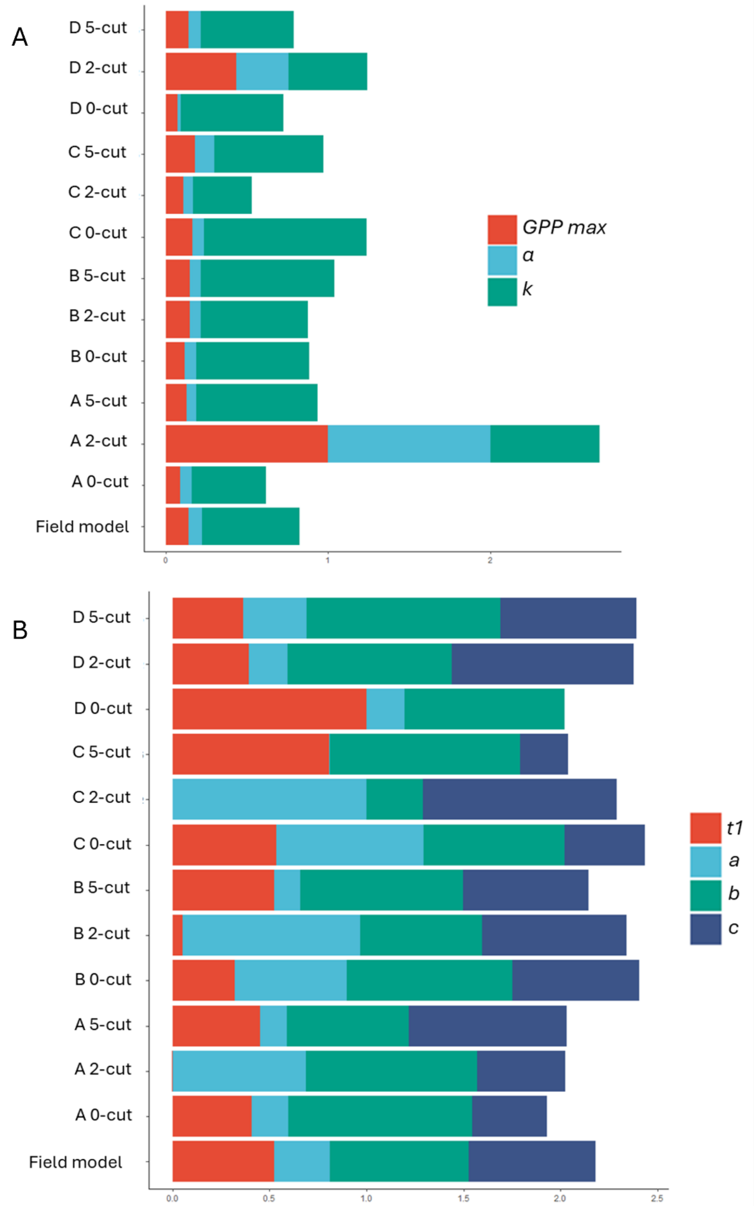

Figure 5. Variability of parameters fitted in R_{eco} model 4 (top) and the GPP model (bottom). Each bar represents a plot, and the bottom bar corresponds to the pooled model. Each color represents a different parameter. Parameter values were normalized i.e. dividing them by the maximum value.

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3.2 Carbon balance

Management had a marginally significant effect on GPP (p value < 0.1), with more negative GPP (highest photosynthesis) in the five-cut treatment (-20.2 ± 0.7 t CO_2-C ha^{-1} yr^{-1}; mean ± SE) and lowest in the 0-cut treatment (-15.5 ± 1.3 t CO_2-C ha^{-1} yr^{-1}) (Table 3). No significant effects of management on R_{eco} (between 22.1 ± 2.5 and 22.4 ± 3.3 t CO_2-C ha^{-1} yr^{-1}; p value = 0.98) and NEE (between 2.2 ± 0.5 and 6.9 ± 2.2 t CO_2-C ha^{-1} yr^{-1}; p value = 0.22) were registered although the NEE of 0-cut was 4.6 t CO_2-C ha^{-1} yr^{-1} higher than the two managed treatments on average. The 2-cut and 5-cut treatments gave similar annual biomass yields (4 ± 0.7 and 4 ± 0.2 t C ha^{-1} yr^{-1}, respectively) leading to similar NECB for all treatments when the exported yields were added to the NEE (between 6.0 ± 0.5 and 6.9 ± 2.2 t CO_2-C ha^{-1} yr^{-1}). Biomass yields of the 2-cut treatment were similar for both harvesting events, but much lower in block A compared to the other blocks, while for the 5-cut treatment yields peaked at the third harvest and were lowest at the fifth. There were less yield differences between blocks for the 5-cut treatment compared to the 2-cut treatment. Block D had the highest yields of both 2-cut and 5-cut treatments.

Although the experimental site looked rather uniform, large differences were seen between blocks, especially for Reco and NEE, the latter with coefficients of variation of 0.56, 0.71, and 0.41, for the 0-cut, 2-cut, and 5-cut, respectively. The lowest R_{eco} was registered in block A, followed by block B, and the highest R_{eco} was in blocks C and D ($p<0.05$) (Table 3, Fig 6). Differences in GPP between blocks were not significant despite the higher GPP leading to lower biomass production in block A. Despite significant differences in R_{eco}, no significant difference in NEE was observed between blocks because the higher R_{eco} was accompanied by lower (more negative) GPP and thus higher biomass production. However, NECB was marginally different (p value < 0.1) between blocks, with lowest NECB in block A, followed

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by block B, and highest in block C and D indicating that within field heterogeneity overrides treatments. Figure 7 shows that the cumulative NEE grew faster in blocks C and D than in blocks A and B leading to approximately eight times higher annual NEE in blocks C and D compared to block A for the 0-cut treatment.

Cumulated methane emissions averaged 113 kg CH_4 ha^{-1} yr^{-1} for the studied year and varied primarily by block with less emissions at block C and largest emissions at block A (Table 4 and Fig. A2). Methane had no significant correlations with nutrients (Fig. A1), except NH_4, which had a negative correlation with CH_4. CH_4 also had a positive correlation with R_{eco} and Ts but no significant correlation with WTD.

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372 Table 3. Cumulated CO_2-C emission for the four studied blocks and harvest treatments during
373 the study year.

Block	Treatment	R_{eco}	GPP	NEE	Yield	NECB
		t CO_2-C ha^{-1} yr^{-1}	t CO_2-C ha^{-1} yr^{-1}	t CO_2-C ha^{-1} yr^{-1}	t C ha^{-1} yr^{-1}	t C ha^{-1} yr^{-1}
A	0-cut	15.4	-14.2	1.2	NA	1.2
B		18.6	-13	5.6	NA	5.6
C		26.2	-16	10.2	NA	10.2
D		29.4	-18.9	10.6	NA	10.6
Mean ± SE		22.4 ± 3.3	-15.5 ± 1.3	6.9 ± 2.2	NA	6.9 ± 2.2
A	2-cut	14.9	-15.3	-0.4	1.9	1.5
B		23.6	-20.8	2.8	4.5	7.3
C		26.4	-22	4.3	4.6	9
D		23.7	-20.6	3.1	5	8.1
Mean ± SE		22.1 ± 2.5	-19.7 ± 1.5	2.5 ± 1	4.0 ± 0.7	6.5 ± 1.7
A	5-cut	20.6	-18.5	2.2	3.5	5.6
B		21	-20.2	0.8	3.9	4.7
C		23.7	-20.4	3.3	3.5	6.8
D		24.3	-21.9	2.4	4.5	6.9
Mean ± SE		22.4 ± 0.9	-20.2 ± 0.7	2.2 ± 0.5	3.8 ± 0.2	6.0 ± 0.5

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375 R_{eco} is ecosystem respiration, GPP is gross primary productivity, NEE is net ecosystem
376 exchange, and NECB is net ecosystem carbon balance (NEE + yield).

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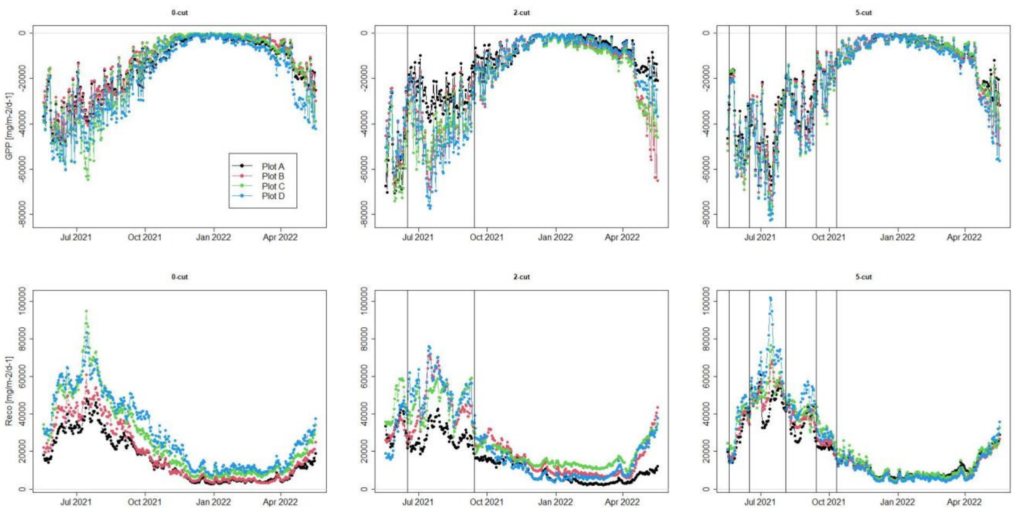

Figure 6. Modelled daily gross primary productivity (GPP) (top) and ecosystem respiration (R_{eco}) (bottom) for the three management treatments (0-cut, 2-cut, and 5-cut). Colors indicate the four block replicates. Vertical lines are harvesting events.

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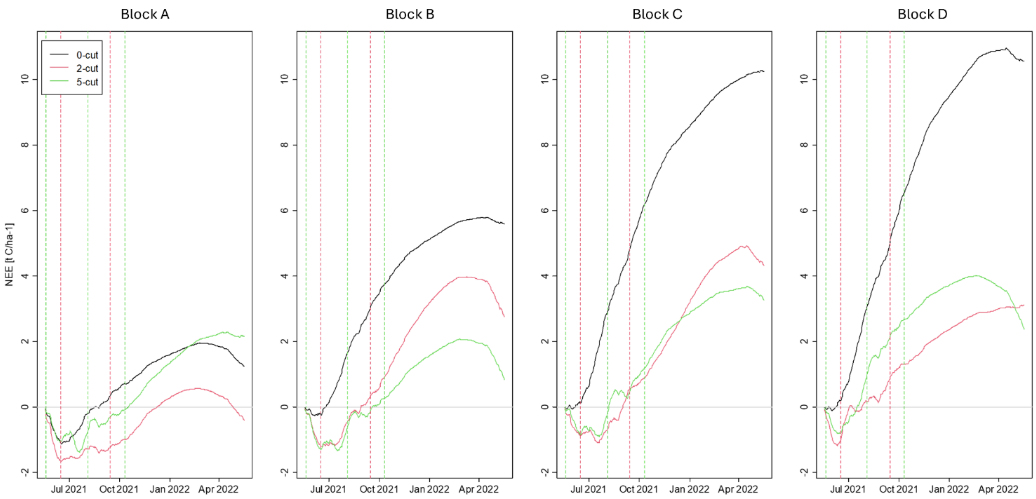

Figure 7. Cummulative net ecosystem exchange for the four studied blocks and three harvest treatments. Black line is the 0-cut, red line is the 2-cut, and green line is the 5-cut treatment. Vertical green dashed lines are harvest events for only the 5-cut treatment while red dashed lines are harvest events for both the 2-cut and 5-cut treatments.

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Table 4. Cumulated methane emissions for the blocks (A, B, C, D) and harvest treatments 0-cut, 2-cut, and 5-cut during the study year.

Block	Treatment	CH_4 emissions
		kg CH_4 ha^{-1}
A	0-cut	200.2
	2-cut	157.5
	5-cut	125.7
	mean ± SD	161.1 ± 30.5
B	0-cut	124.0
	2-cut	129.0
	5-cut	99.4
	mean ± SD	117.5 ± 12.9
C	0-cut	35.7
	2-cut	40.1
	5-cut	73.7
	mean ± SD	49.8 ± 17
D	0-cut	114.5
	2-cut	190.0
	5-cut	67.6
	mean ± SD	124.0 ± 50.4
Total mean		113.1

3.3 Sensitivity analysis using WTD with different temporal resolution

Using annual mean WTD and annual mean Ts as input data for model 4 instead of hourly values, while keeping hourly RVI, underestimated R_{eco} between 9 to 26% for all plots with an average of 18% (Fig 8) (Table A4). On the other hand, using the annual mean WTD along with hourly Ts and RVI generally followed similar trends in R_{eco} as using hourly WTD as input data, but high emission events were slightly underestimated resulting in an underestimation that ranged between 0 and 10% with an average of 5% for all plots when compared to the model using hourly WTD, Ts, and RVI (Fig 8) (Table A4). If these R_{eco} values would be included in the CO_2 budget, this would result in a total mean NECB of 2.6 and 5.3 t C ha^{-1} yr^{-1}, respectively.

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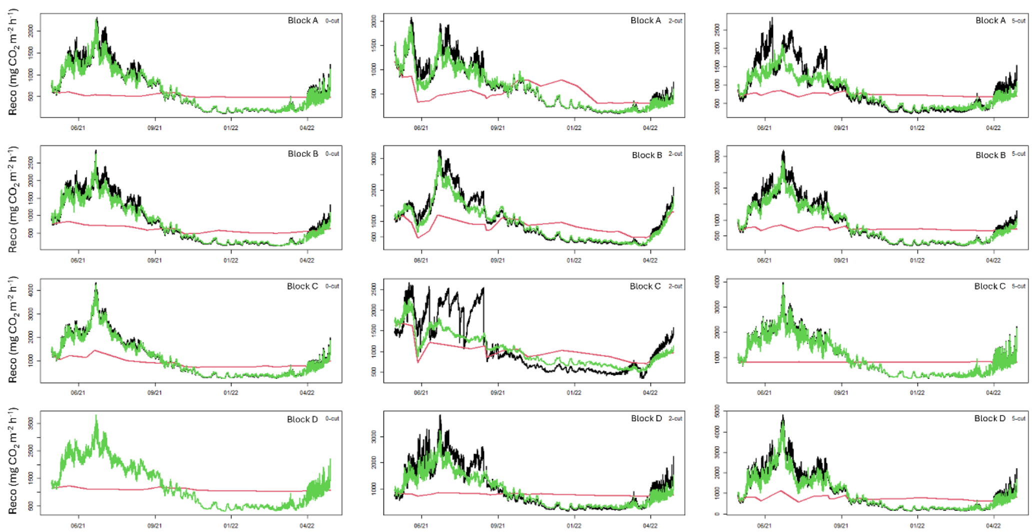

Figure 8. Sensitivity of ecosystem respiration (R_{eco}) modelled for all plots to the data frequency of water table depth (WTD). Black lines represent R_{eco} modelled with hourly WTD, soil temperature (Ts), and RVI, green line represents R_{eco} modelled with mean annual WTD, hourly Ts, and hourly RVI, and red line represents R_{eco} modelled with mean annual WTD, mean annual Ts, and hourly RVI.

3.4 Water chemistry

The PCA described in total 67.8 % of the variance in data by the first three principal components. PC1 and PC2 explained 29.9 and 24.7% of the variability in the data, respectively, while PC3 explained 13.4% of the variability (Fig 4). The PC1 VS PC2 plot shows clustering of the data with blocks D and A having the largest difference. PC1 describes the pore water nutrients, which are positively correlated with each other, and significance of correlations are presented in Figure A1. This shows that WTD had positive correlations with Fe, TOC and DOC and negative correlations with NH_4 and TDP, while Ts had negative correlations with all nutrients except NH_4 and TDP. Predominant correlations of nutrients with R_{eco} were negative and positive with GPP and NEE, respectively.

Comparisons of water chemistry parameters between blocks indicated significant differences depending on type of nutrients. Generally, block (A) had the lowest nutrient concentrations,

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while block (D) had the highest nutrient concentrations, with the exception of DOC. The nutrient concentrations at the ditch appeared lower than the concentrations in the soil pore water at the blocks except for the TP and TDP (Table 5). Comparisons between harvest treatments showed that 2 and 5-cut treatments had higher N and Fe concentrations than the 0-cut treatment, while there were no differences in other nutrients (Table 5). Additionally, the interaction between harvest treatment and block was significant for NH_4, electroconductivity, pH, and turbidity.

The linear mixed model (Model 5) indicated that all nutrient concentrations, except NH_4, significantly improved the base R_{eco} model (Table 6), however the effect of TP, TDP, and pH also varied at plot level. For GPP, the addition of nutrients did not improve the base models, however pH and EC improved model 6 with its effect varying at plot level. The magnitude of model improvement (higher R^2 and lower RMSE) was larger for R_{eco} than for GPP, however, in general the R^2 and RMSE did not change considerably for all nutrients/parameters compared to the base models (Table A5).

Table 5. Mean annual concentrations of water chemistry parameters at block A, B, C, and D, and treatments 0-cut, 2-cut, and 5-cut.

Block	pH	EC	Turbidity	TOC	DOC	TN	TDN	NH_4-N	NO_3-N	TP	TDP	Fe
		mS cm^{-1}	NTU	mg L^{-1}	mg L^{-1}	mg L^{-1}	mg L^{-1}	mg L^{-1}	mg L^{-1}	mg L^{-1}	mg L^{-1}	mg L^{-1}
A	5.61 ± 0.05 (a)	0.19 ± 0.01 (a)	25.4 ± 2.01 (ab)	164 ± 9 (a)	129 ± 7 (a)	14.1 ± 0.9 (a)	12.8 ± 0.8 (a)	1.56 ± 0.25 (a)	4.98 ± 3.18	0.49 ± 0.04 (a)	0.40 ± 0.04 (a)	12.2 ± 0.9 (a)
B	6.40 ± 0.04 (c)	0.34 ± 0.01 (c)	29.6 ± 2.95 (b)	212 ± 7 (b)	160 ± 5 (b)	16.8 ± 0.4 (b)	15.5 ± 0.5 (b)	1.50 ± 0.15 (a)	1.38 ± 0.58	0.81 ± 0.04 (c)	0.69 ± 0.05 (b)	22.9 ± 1.1 (b)
C	6.22 ± 0.04 (b)	0.34 ± 0.01 (b)	40.3 ± 3.76 (c)	193 ± 10 (b)	135 ± 6 (ab)	18.6 ± 0.9 (b)	16.2 ± 0.7 (b)	3.34 ± 0.29 (b)	2.97 ± 1.49	0.68 ± 0.04 (b)	0.50 ± 0.03 (a)	19.0 ± 1.6 (b)
D	6.25 ± 0.04 (b)	0.32 ± 0.01 (b)	26.7 ± 3.85 (a)	209 ± 16 (b)	137 ± 8 (a)	19.6 ± 1.2 (b)	18.9 ± 1.2 (b)	2.95 ± 0.24 (b)	3.58 ± 1.90	1.07 ± 0.08 (c)	0.91 ± 0.08 (b)	36.3 ± 3.6 (c)
ditch	6.65 ± 0.07	0.32 ± 0.01	41.9 ± 32.9	66 ± 8	42 ± 3	7.2 ± 1.8	4.6 ± 0.3	1.2 ± 0.2	1.09 ± 0.21	1.13 ± 0.23	0.93 ± 0.2	3.9 ± 1.1
Treatment												
0	6.13 ± 0.05 (b)	0.26 ± 0.01 (a)	27.3 ± 2.4	191 ± 9	137 ± 5	16.0 ± 0.8 (a)	14.5 ± 0.7 (a)	1.96 ± 0.16 (a)	0.15 ± 0.03	0.83 ± 0.06	0.63 ± 0.05	20.3 ± 1.9 (a)
2	6.04 ± 0.05 (a)	0.31 ± 0.01 (b)	33.3 ± 3.0	189 ± 10	136 ± 6	18.5 ± 0.9 (b)	16.9 ± 0.9 (b)	2.69 ± 0.30 (b)	7.49 ± 2.64	0.71 ± 0.04	0.59 ± 0.05	23.3 ± 1.9 (b)
5	6.20 ± 0.04 (b)	0.33 ± 0.01 (c)	30.5 ± 3.1	203 ± 10	148 ± 6	17.3 ± 0.7 (ab)	16.1 ± 0.7 (ab)	2.36 ± 0.18 (ab)	1.87 ± 0.49	0.76 ± 0.05	0.63 ± 0.05	24.3 ± 2.2 (ab)

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Total organic carbon (TOC), dissolved organic carbon (DOC), total nitrogen (TN), total dissolved nitrogen (TDN), ammonia (NH_4-N), nitrate (NO_3-N), total phosphorus (TP), total dissolved phosphorus (TDP), electrical conductivity (EC). Values are means ± standard error. Letters in parenthesis indicate significant differences between Blocks (top) and harvest treatments (bottom). The ditch was not included in statistical comparisons. No comparisons were performed with NO_3 due to insufficient data.

Table 6. Effect of soil pore water chemistry parameters on ecosystem respiration (R_{eco}, model 5) and gross primary productivity (GPP, model 6).

W.C. Parameter	R_{eco} effect	GPP effect
TOC	Sig	N
DOC	Sig	N
TN	Sig	N
TDN	Sig	N
NH_4	N	N
TP	Sig*	N
TDP	Sig*	N
Fe	Sig*	N
pH	Sig*	Sig*
Turbidity	N	N
EC	N	Sig*

Sig indicates significant improvement of the models by individually adding water chemistry parameters. Sig* indicates a significant effect that varied between harvest treatments. Water chemistry parameters included total organic carbon (TOC), dissolved organic carbon (DOC), total nitrogen (TN), total dissolved nitrogen (TDN), total phosphorus (TP), total dissolved phosphorus (TDP), and electrical conductivity (EC).

4. Discussion

4.1 Management effect on C emissions

Comparison of results from this study to previous flux measurements on managed Danish peatlands presented by Koch et al. (2023) shows that the total mean CO2-C emissions (NECB) from this study (6.5 t CO_2-C ha^{-1} yr^{-1}) are larger than emissions from other Danish organic soils at similar WTD (between 0 and 2.5 t CO_2-C ha^{-1} yr^{-1}; Koch et al., 2023). Similarly, our NECB is larger than emissions from peatlands under drained and rewetted conditions at similar WTD from both Germany (between -1.0 t CO_2-C ha^{-1} yr^{-1} and 1.5 t CO_2-

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C ha^{-1} yr^{-1}; Tiemeyer et al., 2020) and the UK (between -2.0 t CO_2-C ha^{-1} yr^{-1} and 0.8 t CO_2-C ha^{-1} yr^{-1}; Evans et al., 2021). Our NECB results are closer to the lower range of emissions from drained agricultural peatland presented by Koch et al. (2023). Nielsen et al. (2024) reported the effect of management on GHG emissions from 2020 to 2021 at the same study site as reported here, and found a higher mean NECB of 9.4 t CO_2-C ha^{-1} yr^{-1} at the slightly lower mean annual WTD of -10 cm. Although mean annual WTD increased only 2 cm, blocking of the drainage ditches resulted in considerably higher WTD during summer 2021 envisaged by temporary flooded conditions which could explain the lower NECB in 2021-22 compared to 2020-21 (Fig 2A). Other studies have also shown a delay in reaching carbon neutral conditions despite drainage being stopped (Hemes et al., 2019; Kreyling et al., 2021). For the shallow annual mean WTD registered at our study site we expected lower CO_2 emission according to IPCC Tier 1 emission factors. However, here R_{eco} is likely driven by the dynamic interaction of a drop in WTD during summer coinciding with maximum Ts. This naturally stimulated CO_2 production in the peat and together with plant respiration drove the high annual R_{eco} (Fig 6).

Rewetted nutrient-rich fen peatlands have higher CO_2 emissions compared to low-nutrient ones (Wilson et al., 2016). Management alternatives to reduce emissions from these sites are therefore needed in order to meet emission reduction targets. Paludiculture has been found to effectively reduce emissions from rewetted peatlands (Tanneberger et al., 2020; De Jong et al., 2021; Bockermann et al., 2024). Our results showed that after three years of establishment and management of RCG, NECB was not significantly different compared to no management. These results support findings by Nielsen et al. (2024) who found no effect of management on GHG emissions during the second year (2020) after RCG establishment at the study site. The NECB assumes that all harvested biomass is converted to CO_2 when removed from the field. However, if the biomass is considered as a resource potentially

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reducing the use of fossil fuels, comparison of NEE among treatments would also be a relevant measure. Based on NEE, we found a potential emission reduction of 4.5 and 4.7 t CO_2-C ha^{-1} yr^{-1} for the 2 and 5-cut management strategies, respectively, in comparison to no management, but this difference was not significant because of large variation between treatment replicates especially for the 0-cut. Our NEE estimates were lower for all treatments compared to Nielsen et al. (2024). We attribute this reduction in net CO_2 emissions not only to the reduction in biomass production but also to the rewetting process, which lowered heterotrophic peat mineralization. Considering the differences between the studied blocks (treatment replicates), the potential reduction in CO_2 emission was larger in higher emission areas, which in this study were also the areas with higher porewater nutrient concentrations. In high emission areas, we found a potential reduction of up to 8.2 t CO_2-C ha^{-1} yr^{-1} (block D, 5-cut treatment) based on NEE and 3.7 t CO_2-C ha^{-1} yr^{-1} based on NECB. However, in areas of less emissions (block A in this study), harvest of the biomass could not be recommended as no benefit was seen. These results stress the importance of acknowledging peatland heterogeneity in rewetting projects to maximize emission reductions.

A life cycle assessment of RCG on fen peatlands by Thers et al. (2023) showed that fuel consumption during harvesting can make up a considerable amount of GHG emissions associated to management. Since no considerable difference in yields were found between the 2-cut and 5-cut treatments, and a progressive decline was seen after the third harvest of the 5-cut treatment, we would recommend the 2-cut management for RCG in peatlands such as the study site to maximize harvest efficiency and to minimize disturbance to the peatland. Although yields of 2021 (8.9 and 8.6 t DM ha^{-1}) (Table A1) were acceptable they were considerably lower compared to 2019 yields (15.6 and 14.9 t DM ha^{-1}) (Nielsen et al., 2021) and to 2020 yields (12.7 and 13.8 t DM ha^{-1}) (Nielsen et al., 2023a) for the 2-cut and the 5-cut, respectively. The amount of N removed in the harvested biomass was on average 206 kg

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N ha^{-1} and slightly lower in the 2-cut compared to the 5-cut (Table A2), therefore, the same amount of N applied as fertilizer was removed at harvest. However, we found generally higher concentrations of N forms in pore water at the 2 and 5-cut treatments compared to the 0-cut treatment. A complete assessment of the N balance would help to determine the full environmental benefit of RCG as paludiculture.

Mean CH_4 emissions from this study were within the range of emissions from other Danish and German peatlands reported by Koch et al. (2023) and Tiemeyer et al. (2020) and no treatment effect was apparent. We found that CH_4 emissions contributed 11.3% to total net mean NECB expressed as CO_2e (using GWP = 27 for CH_4). Peatland rewetting is expected to reduce CO_2 emissions while simultaneously increasing CH_4 emissions (Abdalla et al., 2016; Darusman et al., 2023). Thus, further monitoring of CH_4 emissions would be needed as rewetting progresses at the study site.

4.2 Peatland heterogeneity

Even though the studied area was relatively small (3.9 ha) and appeared to be uniform, we found differences in CO_2 emissions and porewater nutrients among the studied blocks, and preliminary peat chemistry data (Table 1) indicated some differences in pH, organic matter content, and TC among the studied blocks which might be due to the peat forming process. Peatlands can be heterogeneous due to topography, groundwater flow, and vegetational variability, which can produce differences within peatlands in GHG emissions, pore water nutrient concentrations, and microbial communities (Arsenault et al., 2019; Chronakova et al., 2019; Kou et al., 2020). Fen peatlands in particular, have considerable heterogeneity with variable rates of peat and C accumulation (Piilo et al., 2020), and R_{eco} (Juszcak et al., 2013). Mashadi et al. (2024) found an increasing degree of peat decomposition at the study site approaching the stream, therefore, higher nutrient concentrations at blocks closer to the

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stream could be explained by higher peat decomposition and organic matter mineralization at this area. Heterogeneity at the study site could also be seen by considerable variability in values of the fitted parameters of the R_{eco} and GPP models (Fig 5). Pooling all data to obtain field R_{eco} and GPP models resulted in lower model efficiencies (Table A6) compared to the approach of modelling each plot separately and led to similar R_{eco}, GPP, and NEE among treatments and blocks (Table A7). Higher model efficiencies and a better representation of CO_2 emissions from rewetted peatlands can be obtained by considering heterogeneity in these estimations.

4.3 Sensitivity to temporal resolution of WTD for prediction of R_{eco}

In previous studies, mean annual WTD have been used as the only predictor for NECB, but not without considerable variation in data points used to build these relationships (Tiemeyer et al., 2020; Evans et al., 2021; Koch et al., 2023). We found that information on Ts, RVI and PAR improved prediction as they have large impact on GPP and R_{eco}. Out of the three R_{eco} models we tested, the combined model including RVI, WTD and Ts performed best (model 4). When R_{eco} was estimated by models 2 or 3, where either RVI or WTD was omitted the annual R_{eco} and thus NECB was underestimated by 0.6 and 0.2 t C ha^{-1} yr^{-1}, respectively. Therefore, model selection is important to accurately estimate CO_2 emissions from peatlands. The other two models evaluated (models 2 and 3) included Ts as explanatory variable. Temperature is a major soil respiration driver (Silvola et al., 1996; Lafleur et al., 2005; Rigney et al., 2018). Higher soil temperatures increase microbial activity and soil respiration. Soil respiration response to temperature changes, however, depend also on water table and soil moisture (Silvola et al., 1996; Lafleur et al., 2005).

In this study, Ts captured major trends in R_{eco}. This can be seen by the importance of the fitted Ts parameter (*b*, model 4) (Fig 5) and by results shown in Figure 8, in which hourly Ts

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along with mean annual WTD captured most R_{eco} trends, However, this model underestimates R_{eco} by an average of 5%, which would be equivalent to an NECB underestimation of 1.2 t C ha^{-1} yr^{-1} compared to the model with hourly WTD and hourly Ts. The use of mean annual WTD and mean annual Ts would result in an even larger NECB underestimation (3.9 t C ha^{-1} yr^{-1}) compared to the hourly model. This underestimation is due to the combined effect of lower WTD and higher Ts during summer, dominates R_{eco}, and this is not captured when mean WTD and Ts are used. These results showed the importance of using either high temporal resolution WTD or Ts data, if available, to improve R_{eco} and NECB estimates from rewetting peatlands. The model based on hourly WTD and Ts also improved simulation of R_{eco} peaks (Fig 8), which might be of great importance under extreme weather and climate change conditions. Juszcak et al (2013) also found that the response of R_{eco} to Ts can be influenced by WTD and that models including both WTD and Ts provide a better representation of R_{eco} in heterogeneous peatlands. Emission factors derived from models based on annual mean WTD, such as those currently used for rewetted peatlands would underestimate R_{eco} when applied to peatlands with fluctuating and lower WTD during the warm season. This is an important observation particularly for rewetted peatlands, which might take years to achieve hydrological stability (Kreyling et al., 2021). Improved CO_2 modelling therefore requires information on fluctuating WTD possibly obtained from hydrological modelling if measurement data are unavailable.

4.4 Effect of nutrients in CO_2 emissions

Positive correlations between porewater nutrients suggest common drivers for their release. Concentrations of dissolved organic matter components have been found to correlate with concentrations of metals in Canadian bogs (Bourbonniere, 2009). Peat mineralization has been found to be a major driver of nutrient release from drained peatlands (Cabezas et al., 2013; Haapalehto et al., 2014). Predominantly higher nutrient concentrations at the studied

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blocks compared to the ditch indicate differences between the pore water (measured at the plots) and the groundwater (measured at the ditch), suggesting that peat mineralization is the major pore water nutrient source. Peat nutrient concentrations and pH have been found to be potential indicators for GHG emissions in rewetting peatlands (Nielsen et al., 2023b). We showed that the prediction of R_{eco} was improved when soil pore water chemistry data were included in addition to WTD, RVI and Ts as fixed factors. Although, the magnitude of this improvement was small based on the R^2 increase, it indicated a relation between mineralization and porewater nutrients at the study site. The exact influence of nutrients in R_{eco} should be further investigated. In this study we measured nutrient concentrations but not nutrient load, which is the total mass of a nutrient and can be more informative about the nutrient status of the peatland (Cabezas et al., 2013). Under higher (shallower) WTD, nutrient concentrations can be diluted (Griffiths et al., 2019), Minor differences in WTD between the studied blocks could produce different degree of exposure to incoming water sources and explain lower nutrient concentrations at block A. Positive correlations between WTD and TOC, DOC and Fe could be due to release of DOC accumulated under drained summer conditions and increase in Fe solubility under higher water tables (Haapalehto et al., 2014).

Previous studies have explored variability on water chemistry between and within peatlands (Bourbonniere, 2009; Wood et al., 2016; Arsenault et al., 2018; Griffiths et al., 2019). Nutrient concentrations in peatland's porewater are affected by several factors including water table depth, temperature, peat decomposition degree, and redox (Bourbonniere, 2009; Cabezas et al., 2013; Haapalehto et al., 2014; Wood et al., 2016). For this study, WTD was generally lower in blocks C and D (Fig 2B). Malinowski et al. (2015) found that the area where block A is located is more responsive to changes in the stream water level due to its proximity to the drainage ditch, which might have caused higher WTD at this block. Additionally, differences in mobile porosity at the study site might have made some areas more prone to be affected by changes in WTD than others (Mashadi et al., 2024). The minor differences found in WTD might increase peat mineralization in drier blocks resulting in higher DOC and N concentrations (Arsenault et al., 2018; Haapalehto et al., 2014; Wood et al., 2016). Higher mineralization from block D was also evidenced by higher R_{eco} found in this block. Higher plant productivity and fresh decomposable organic matter contributes to higher nutrients found in rewetted peatlands (Haapalehto et al., 2014), which could explain higher N concentrations found in blocks C and D. This is also supported by marginally higher ($p < 0.1$) NECB found in these blocks compared to blocks A and B. A feedback mechanism by which higher mineralization and nutrient release enhances plant productivity, which in turn increases fresh organic matter inputs into the soil and further nutrient releases could drive high nutrient concentrations in poorly drained fen peatlands such as this one.

4.5 Considerations for the potential use of RCG harvested biomass

In order to reestablish the C sink function of rewetted peatlands, peat formation would need to be reestablished, however, reaching this state may take decades (Kreyling et al., 2021). Through replacing the use of fossil fuels, paludiculture provides an opportunity for achieving an indirect emission reduction, since harvested biomass C makes out a considerable amount of GHG emissions from cultivated RCG in fen peatlands (Thers et al., 2023). The end use of the harvested biomass is key to achieve the potential GHG mitigation. Reed canary grass grown in wet Danish fen peatlands was shown suitable for protein extraction as supplement in the diets of monogastric animals and side streams or all the harvested biomass could be used for biogas production thereby replacing fossil fuels (Kandel et al., 2013; Nielsen et al. 2021; Nielsen et al., 2023a). The feasibility of using biomass from reed canary grass to offset fossil fuels would depend on the development of non-invasive harvesting techniques, the identification of viable and economically suitable uses for this biomass, and the establishment of markets and infrastructure for its processing.

5. Conclusion

We found that harvesting moderately fertilized RCG in the third production year did not increase net C emissions significantly in poorly drained fen peatlands compared to no management. Additionally, the NECB was reduced further under management compared to previous years as rewetting progressed, and a biomass resource that potentially could reduce GHG emission elsewhere depending on the end use was produced. Considering the heterogeneity of the field, results also indicated that harvest of the biomass only reduced net C fluxes at nutrient rich areas. At relatively nutrient poor areas it seems more advantageous to leave the grass without management. Paludiculture and management of RCG in rewetting fen peatlands, therefore, offers an alternative that could be particularly beneficial in high nutrient rich areas. We found that differences in annual NECB were highly influenced by R_{eco}, and that R_{eco} was best modelled by daily data on RVI, WTD and Ts, with R_{eco} being underestimated when the mean WTD was used instead of hourly values, indicating that temporal variability in WTD should be considered in establishing emission factors for rewetted fen peatlands. Differences in porewater nutrient concentrations were able to further improve prediction of R_{eco} based on a statistical model. As more nutrients could be related to higher CO_2 emissions, we suggest a feedback mechanism driving the mineralization, nutrient release, biomass production and peatland heterogeneity. Further research and the establishment of infrastructure and markets for harvested biomass would improve the prospects of paludiculture in rewetted peatlands.

Competing interests

The authors declare that they have no conflict of interest.

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Appendix A

Table A1. Biomass yields for each harvest event.

treatment	block	Yield per harvest event (t DM ha^{-1})					
		20-May	16-jun	04-Ago	14-sep	11-Oct	Total
2-cut	A	-	2.8	-	1.4	-	4.2
2-cut	B	-	5.3	-	4.8	-	10.1
2-cut	C	-	4.4	-	5.8	-	10.2
2-cut	D	-	4.6	-	6.5	-	11.1
5-cut	A	1.5	1.5	2.9	1.4	0.5	7.8
5-cut	B	1.0	2.6	3.0	1.7	0.5	8.7
5-cut	C	0.3	1.3	3.5	2.1	0.7	7.8
5-cut	D	1.2	1.9	4.2	2.0	0.7	10.1

Table A2. Total N in harvested biomass per event

treatment	block	total N in biomass per harvest event (kg ha^{-1})					
		20-May	16-jun	04-Ago	14-sep	11-Oct	Total
2-cut	A	-	62	-	31	-	93
2-cut	B	-	104	-	91	-	195
2-cut	C	-	99	-	116	-	215
2-cut	D	-	91	-	113	-	204
5-cut	A	49	38	56	40	21	204
5-cut	B	34	61	65	52	20	233
5-cut	C	13	35	93	74	31	245
5-cut	D	41	47	83	59	27	258

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Table A3. Model evaluation for GPP model. Values are three different indexes of model performance for each block.

Block	Treatment	GPP model		
		R^2	NRMSE	NSE
A	0-cut	0.9	31.1	0.9
	2-cut	0.94	24.2	0.94
	5-cut	0.89	33.9	0.88
B	0-cut	0.91	29.9	0.91
	2-cut	0.96	19.3	0.96
	5-cut	0.94	24.4	0.94
C	0-cut	0.92	27.6	0.92
	2-cut	0.81	43.4	0.81
	5-cut	0.91	30.6	0.91
D	0-cut	0.86	37.5	0.86
	2-cut	0.91	29.3	0.91
	5-cut	0.9	32.4	0.89

A, B, C, and D are the four block replicates, The three harvest treatments at each block are 0, 2, and 5. The three indexes of model evaluation are: R2, normalized root mean square of error (NRMSE), and Nash-Sutcliffe efficiency (NSE).

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Figure A1. Pearson's correlations of water chemistry parameters, Ecosystem respiration (R_{eco}), net ecosystem exchange (NEE), gross primary productivity (GPP), water table depth (WTD), soil temperature at 5 cm depth (T5), ammonia (NH4.N), total nitrogen (TN), total dissolved nitrogen (TDN), total phosphorus (TP), total dissolved phosphorus (TDP), total organic carbon (TOC), and dissolved organic carbon (DOC). * significant at $p < 0.05$, ** significant at $0.01 > p > 0.001$*** significant at $p < 0.00$

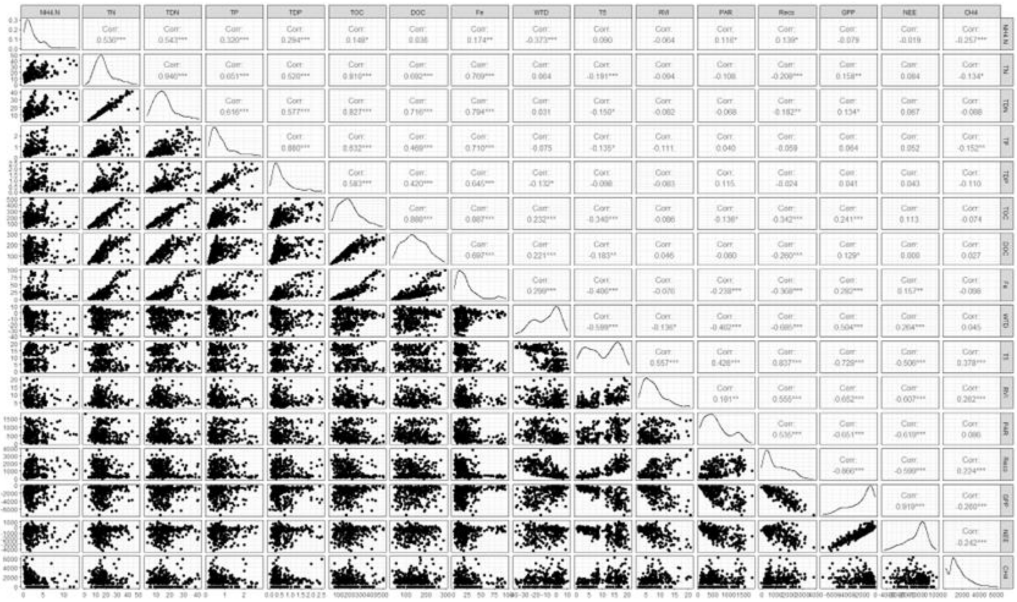

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Figure A2. Time series of methane emissions from studied blocks (different line colors) and from the three harvest treatments (zero cut, two cut, and five cut). Each dot in the lines represents a measurement campaign. CH_4 emissions calculated only under 0% PAR conditions.

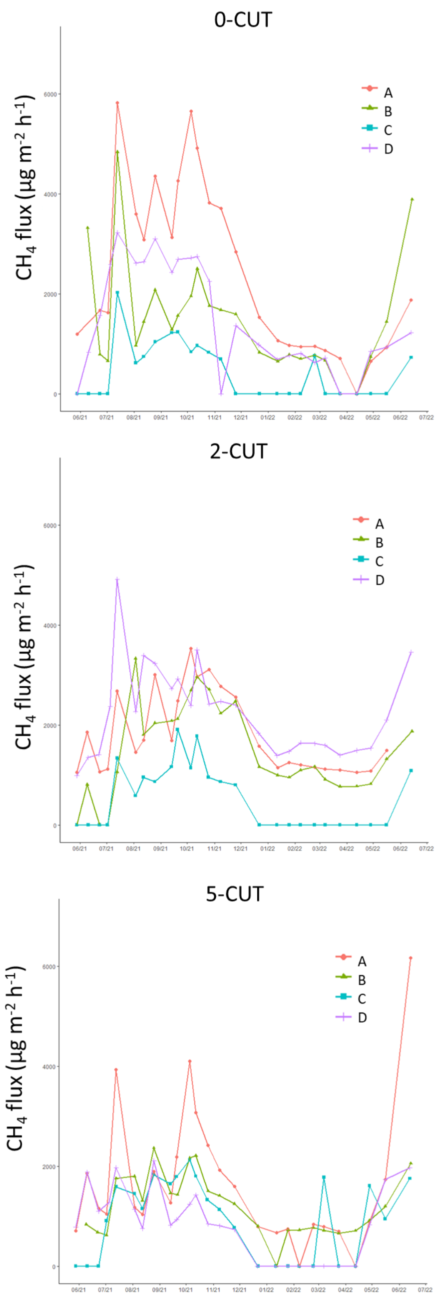

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Table A4. Comparison of annual R_{eco} estimated with models 4, 2 and 3, which use hourly data on Ts, WTD and RVI, model 4 using either mean annual WTD and Ts (M Model), and model 4 using mean annual WTD and hourly Ts (MH model), the latter two models including hourly RVI data.

Block	**Treatment**	**Model 4**	**Model 2**	**Model 3**	**M model**	**MH model**
		t CO_2-C ha^{-1} yr^{-1}				
A	0	15.4	14.9	15.4	12.2	14.8
B		18.6	18.4	18.9	14.4	17.7
C		26.2	26.1	25.6	21.6	25.8
D		29.4	29.4	31	25.9	29.4
Average ± SE		22.4 ± 3.3	22.2 ± 3.4	22.7 ± 3.5	18.5 ± 3.2	21.9 ± 3.4
A	2	14.9	14.5	15.1	12.3	13.9
B		23.6	23.4	23.6	20.5	22.4
C		26.4	25.7	26	24.1	24.4
D		23.7	22.7	23	18.7	21.4
Average ± SE		22.1 ± 2.5	21.6 ± 2.4	21.9 ± 2.4	18.9 ± 2.5	20.6 ± 2.3
A	5	20.6	18.6	19.3	17.4	18.6
B		21	20.8	20.5	17.0	19.7
C		23.7	23.6	23.4	19.8	23.4
D		24.3	22.9	23.4	17.9	22.6
Average ± SE		22.4 ± 0.9	21.5 ± 1.1	21.7 ± 1	18 ± 0.6	21.1 ± 1.1

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Table A5. Total organic carbon (TOC), dissolved organic carbon (DOC), total nitrogen (TN), total dissolved nitrogen (TDN), total phosphorus (TP), total dissolved phosphorus (TDP), Turbidity (NTU), electrical conductivity (EC). If base model did not improve by adding the water chemistry parameters, R^2 and RMSE are not shown.

Parameter	**Reco models**				**GPP models**			
	Base R^2	Improved R^2	Base RMSE	Improved RMSE	Base R^2	Improved R^2	Base RMSE	Improved RMSE
TOC	0.863	0.873	243	226	-	-	-	-
DOC	0.863	0.871	242	228	-	-	-	-
TN	0.863	0.870	244	229	-	-	-	-
TDN	0.864	0.876	242	224	-	-	-	-
NH_4	-	-	-	-	-	-	-	-
TP	0.867	0.871	241	231	-	-	-	-
TDP	0.862	0.867	244	229	-	-	-	-
FE	0.863	0.878	242	225	-	-	-	-
pH	0.863	0.868	243	239	0.832	0.839	645	628
NTU	-	-	-	-	-	-	-	-
EC	-	-	-	-	0.832	0.839	643	624

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Table A6. Model evaluation of R_{eco} and GPP models using all data pooled and modelling all blocks and harvest treatments all together "field model"

R_{eco} model	R^2	0.78
	NRMSE	46.6
	NSE	0.78
	AIC c	14223.49
GPP model	R^2	0.88
	NRMSE	34.2
	NSE	0.88

The four indexes of model evaluation are: R2, normalized root mean square of error (NRMSE), Nash-Sutcliffe efficiency (NSE), and corrected Akaike Information Criteria.

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Table A7. Carbon budget results obtained by using all data pooled and modelling all blocks and harvest treatments all together to obtain field models of R_{eco} and GPP.

Block	**Treatment**	**Reco** t CO_2-C ha^{-1} yr^{-1}	**GPP** t CO_2-C ha^{-1} yr^{-1}	**NEE** t CO_2-C ha^{-1} yr^{-1}	**Yield** t C ha^{-1} yr^{-1}	**NECB** t C ha^{-1} yr^{-1}
A	0-cut	21.1	-16.9	4.2	NA	4.2
B		18.8	-15.6	3.2	NA	3.2
C		21.6	-16.6	5.0	NA	5.0
D		23.0	-19.2	3.8	NA	3.8
Mean ± SE		21.1 ± 1.6	-17.1 ± 1.3	4.1 ± 0.7	NA	4.1 ± 0.7
A	2-cut	21.9	-17.5	4.4	1.9	6.3
B		22.4	-19.3	3.1	4.5	7.7
C		23.7	-18.4	5.3	4.6	10.0
D		22.1	-16.6	5.5	5.0	10.6
Mean ± SE		22.6 ± 0.7	-17.9 ± 1	4.6 ± 1	4 ± 0.7	8.6 ± 1.7
A	5-cut	23.9	-19.4	4.4	3.5	7.9
B		23.7	-20.8	2.9	3.9	6.7
C		25.7	-20.7	5.0	3.5	8.5
D		23.8	-20.3	3.5	4.5	8.0
Mean ± SE		24.3 ± 0.8	-20.3 ± 0.6	3.9 ± 0.8	3.8 ± 0.2	7.8 ± 0.7

R_{eco} is ecosystem respiration, GPP is gross primary productivity, NEE is net ecosystem exchange, and NECB is net ecosystem carbon balance (NEE + yield).