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Implementation of regenerative ditch borders in Dutch peat meadows: effects on soil CO_2 fluxes and potential carbon trade-offs

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Abstract

Drained peatlands account for 3 % of the Dutch national greenhouse gas (GHG) emissions. Topsoil removal (TSR; which leads to removal of built-up nutrients and labile soil carbon) in combination with rewetting (groundwater table management) in drained peatlands is often proposed as a restoration measure effective in reducing carbon dioxide (CO_2) emissions, yet often politically challenging to implement. TSR is however currently applied at a smaller scale during the implementation of regenerative ditch borders (RDBs), but its effectiveness in reducing CO_2 remains uncertain. We investigated the effects of RDBs on soil CO_2 emissions in comparison to conventional borders (CDBs) in a year-round field survey (June 2023 – March 2024) in a lowland peat agroecosystem. Soil respiration was measured at four distances from the water's edge (40, 80, 360 and 640 cm) over a ten-month period and used to fit statistical models with the predictors soil temperature, soil moisture content and exposed carbon (a variable integrating profile soil carbon density and groundwater level). The resulting model was used to calculate annual soil respiration, and to estimate the payback time (no. of years for net negative effects on cumulative CO_2 emissions) of the removed carbon. Spatiotemporal variation in soil respiration was mostly explained by exposed carbon and soil temperature (32 % and 30 %, respectively). Soil respiration peaked at 65 % soil moisture content. At distances 40 and 80 cm reduced soil respiration in RDBs in spring and summer was driven by lower amounts of exposed carbon, while at 360 and 640 cm in RDBs higher soil temperatures and soil moisture content mostly counteracted this effect. Model-based annual soil respiration was 17 % lower in RDBs in comparison to CDBs. If it is assumed that all soil carbon removed during RDB construction is mineralized, the payback time can exceed 100 years. While RDBs can promote reductions in soil CO_2 emissions and therefore more sustainable peatland-adapted agriculture, potential emissions from excavated carbon should be accounted for.

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1 Introduction

Globally peatlands function as an important carbon stock. They cover only 3 % of the world's land surface, yet store an estimated 547 Gt of carbon, predominantly in peatlands located in the Northern Hemisphere (Yu et al., 2010). In peat ecosystems, high water tables create anoxic conditions which have facilitated organic matter accumulation since the Last Glacial Maximum (Gorham, 1991; Yu et al., 2010; Leifeld & Menichetti, 2018). Over the last decades, human-induced land use changes - most importantly land reclamation for agricultural purposes, or mining of peat for fuel - have significantly impacted peatlands, threatening their status as carbon sinks (Joosten, 2009; Tanneberger et al., 2021). Currently, over 65 million ha of peat soils are classified as degraded, with drainage being the main driver (Joosten, 2015). In Northwest Europe, peatlands have been drained for agricultural purposes for centuries to allow cattle and heavy machinery on the land, and to ensure high grassland productivity (van de Ven, 1993; Wereld Natuur Fonds, 2020). However, drainage enables oxygen intrusion into the soil, which enhances aerobic microbial decomposition rates in the unsaturated peat zone, and therefore increases CO_2 emissions into the atmosphere and induces land subsidence (Brouns et al., 2014; Erkens et al., 2016; Joosten, 2009). Ongoing processes of peat degradation therefore have major implications for the soil carbon cycle, resulting in calls for peatland management actions that minimize or reverse the negative consequence of human activity (Chimner et al., 2017; Humpenöder et al., 2020; Zak & McInnes, 2022). Improving our process understanding and prediction accuracy of CO_2 emissions from drained peatlands is important for accurate carbon accounting and assessment of the carbon-cycle impact of these proposed management measures.

In peat soils the key process driving soil CO_2 emissions is microbe-mediated enzymatic breakdown of organic matter which is mostly driven by reaction kinetics, availability of electron acceptors and the biochemical composition of the substrate (Findlay, 2021). Environmental condition affect the reaction kinetics of soil carbon decomposition, with increasing soil temperatures causing exponentially increasing soil CO_2 emissions due to enhanced microbial activity and reaction rates (Davidson & Janssens, 2006; Kucera & Kirkham, 1971). Oxygen is the electron acceptor in the process of aerobic peat

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decomposition, and it's availability in the soil therefore strongly determines peat oxidation and soil CO_2 emissions (Boonman et al., 2024). The biogeochemical quality of the substrate is mostly driven by the chemical composition of the litter of the main peat-building plant species (e.g. *Alnus glutinosa*, *Carex* spp., *Phragmites australis*, or *Sphagnum* spp. in temperate and boreal-zone peatlands), which can influence inherent decomposition rates (Tolunay et al., 2024). These pore-scale processes can be translated into field-scale drivers of soil CO_2 emissions, such as soil moisture content, water table depth, carbon availability and seasonality. When soil moisture content is low it limits the supply of substrates and creates osmotic stress for soil microbes and when soil moisture is high the diffusion of oxygen becomes limited (Skopp et al., 1990; Schjønning et al., 2003; Manzoni et al., 2012; Moyano et al., 2013). Therefore, soil respiration shows a parabolic relationship with soil moisture content with reduced soil respiration rates under both low and high soil moisture levels (Moyano et al., 2013; Linn & Doran, 1984). Water table depth regulates soil moisture and oxygen availability, therefore indirectly determining CO_2 emissions (Evans et al., 2021; Tiemeyer et al., 2020). Carbon availability is the result of past hydrological conditions, the dominant peat-forming plant species, plant production, and climate (Loisel et al., 2014). This results in spatial variation in soil carbon density (i.e. quantity of carbon per peat volume), which in turn contributes to spatial variation in soil CO_2 emissions. This interdependency means it can be useful to integrate water table depth and carbon availability into a single predictor variable, exposed carbon (amount of carbon above groundwater table), which indirectly determines CO_2 emissions (Aben et al., 2024). In addition to these, seasonal variation in temperature and water table depth also determine soil CO_2 emissions (Armstrong et al., 2015). Consequently, combining multiple drivers of field-scale emissions is necessary for accurate modelling of the effects of management measures and spatiotemporal variation in peatland soil CO_2 emissions, as changes in management will affect these field-scale drivers.

In the Netherlands, peat soils are responsible for an estimated ± 25 % of the GHG emissions of the agricultural sector, underlining the need for management measures to reduce these emissions (Baren et al., 2024). An often proposed management measure is the combination of topsoil removal (TSR) with rewetting (groundwater table rise) or

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paludiculture (wet agriculture) where TSR, the removal of the topsoil varying between 5 cm and 60 cm in depth, is effective in removing labile carbon and nutrients that have typically accumulated as a consequence of long-term drainage (Zak et al., 2010; Brouns et al., 2014). This removal reduces CH_4 emissions that usually follow a rise in groundwater table and eutrophication of surrounding surface water (Harpenslager et al., 2015; Zak et al., 2017; Huth et al., 2020; Quadra et al., 2023; Käärmelahti et al., 2024; Van Den Berg et al., 2024). Although TSR and rewetting have been proposed as measures for reducing GHG emissions from peatland meadows, their implementation at large scales is currently not feasible in areas used for agricultural production due to the loss of productivity, and resulting economic and political consequences. Nevertheless, in Dutch peatland meadow areas TSR is currently used as part of a set of management practices designed to enhance the biodiversity in the border zones of field border ditches and to improve water quality by increasing nutrient retention in these ditch borders. These practices, also referred to as regenerative ditch borders (or nature-friendly ditch margins), are actively stimulated by Dutch waterboards (regional governing bodies responsible for water management) (van Vossen & Verhagen, 2009). Regenerative ditch borders also have great potential to contribute to CO_2 emission reductions which is currently overlooked. This is important, as a negative side effect of TSR occurs when the removed soil mineralizes and releases CO_2 when placed under aerobic conditions (Van Den Berg et al., 2024). The resulting accrued carbon debt needs to be accounted for, as it can substantially delay net carbon emission reductions and the realization of climate benefits (Huth et al., 2022; Van Den Berg et al., 2024). It is therefore important to estimate the effectiveness of TSR in reducing CO_2 emissions, as TSR is a widely proposed and implemented management measure with possible carbon trade-offs.

Implementing TSR in ditch borders will cause changes in local abiotic conditions in the peat soil, as excavation creates a gradual slope that extends from five meters into the field and descends to the point where the water's edge is met. The newly created topsoil is therefore positioned closer to the water table (i.e. lower elevation level) than topsoil pre-TSR leading to increased frequency of water saturated conditions in these upper soil layers. Especially during the dry summer periods, when groundwater levels are lower, the surface layers of soils in these ditch borders remain wetter in comparison to soils in

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conventional ditch borders. As temperature effects are more profound when peat soils are drier (Kechavarzi et al., 2010), wetter conditions during summer can therefore potentially buffer temperature effects on soil respiration (Fernández-Pascual & Correia-Álvarez, 2021). In addition to these changes in soil temperature and moisture conditions, excavation also directly reduces the total amount of carbon exposed to oxygen under a given surface area. All these effects of TSR are co-occurring and influenced by seasonality, such that the mechanisms expected to lead to CO_2 emission reduction are intertwined. It's therefore challenging to attribute the effectiveness of regenerative ditch borders to a single parameter (Armstrong et al., 2015). This is of particular importance, given that parameter effects may change in magnitude over the seasons and with respect to one another. While parameters can explain variation in soil respiration independently of seasonality, the extent to which other parameters can explain variation may change over time. Understanding of how these parameters change and affect CO_2 emissions will enhance our ability to predict the emission mitigation potential of regenerative ditch borders and a range of other potential management measures that aim to reduce CO_2 emissions from drained peatlands.

In this study we aimed to understand how ditch border management affects soil temperature, soil moisture content, exposed carbon densities and consequently how this practice influences CO_2 emissions from soil respiration in lowland managed peat meadows in the Netherlands. We established a year-round field survey in which we measured soil respiration in conventional vs. topsoil-removed regenerative ditch borders. Measurements were performed along a transect from the water's edge into the agricultural field, covering a representative range of soil conditions. Following the field study, we fit statistical models to assess the effects of soil temperature, moisture, and carbon densities exposed to oxygen on soil respiration under both ditch border management types. We then used the best fitting model to answer the following questions: (1) how do soil temperature, soil moisture content and exposed carbon relate to spatiotemporal variation in soil respiration?; (2) what is the effect of regenerative ditch borders (i.e. TSR) on seasonal and annual modelled soil respiration?; (3) to what extent do soil temperature, soil moisture content, and exposed carbon contribute to differences in soil respiration between conventional and regenerative ditch borders throughout the

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measuring period?; and (4) over what timescale do eventual changes in soil respiration rates compensate for possible carbon debts resulting from TSR?

2 Material & Methods

2.1 Study area

The study area was located in the Vereenigde Binnenpolder, Spaarnwoude, province of North Holland, the Netherlands (52°40'34.1'' N, 5°9'55.4'' E). This minerotrophic peat polder has been described in detail in previous research (Bethe et al., 2025). The dominant land-use of the polder is conventional livestock farming and consists of grasslands separated by ditches. In 1997, a total of 4 ha of multiple regenerative ditch borders (RDB) were constructed by excavating the topsoil layer creating a gentle down-hill slope starting from five meters into the agricultural field and continuing towards the waterside (Fig. 1). After the RDBs were established, management in these borders was adjusted to no manure use, no or delayed mowing, and no biomass harvesting. In the same polder, the remaining ditch borders (hereafter conventional ditch borders - CDB) were subjected to the same management as the adjoining agricultural field apart from a mandatory 1.5 m buffer zone (measured from the water's edge) where no manure use was allowed. This zone was still subject to regular mowing (1-3 times a year) for hay production (Rijksdienst voor Ondernemend Nederland, 2024). In both RDBs and CDBs the agricultural fields were subject to cattle grazing with comparable cattle stocking densities. In RDBs, TSR resulted in substantial vegetational changes (Fig. 1). Close to the water's edge, the vegetation was dominated by *Phragmites australis* with *Mentha aquatica* patches in between and further up the slope a variety of herb and grass species, such as *Lolium perenne* and *Lotus pedunculatus*. In CDBs, the vegetation was dominated by grass species, such as *Lolium perenne*, with some herb species, such as *Ranunculus repens*, growing closer to the water's edge.

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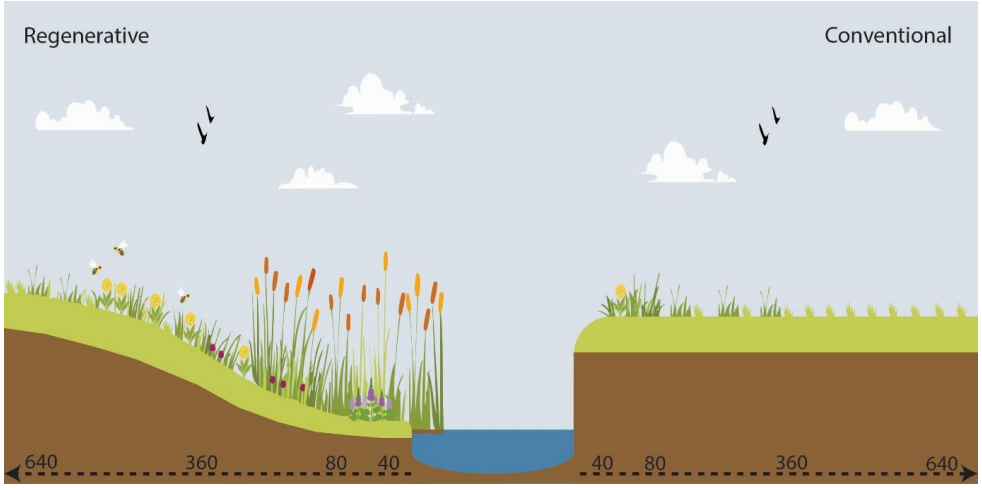

Fig. 1: Schematic comparison between the two ditch border types investigated in this study; a conventional ditch border (right; CDB) and a regenerative ditch border (left; RDB) with a ditch in the middle (blue). In the RDB, a down-hill slope was established by topsoil removal (TSR) from the field towards the water's edge. Distances to the water's edge (see experimental design) 40 cm, 80 cm and 360 cm from the water's edge were therefore located at a lower elevation than corresponding distances in CDBs. Distance 640 cm was located outside the excavation range in RDBs. A buffer zone was located in the CDBs extending 1.5 m into the field from the water's edge. In RDBs, TSR coincided with vegetational changes. Close to the water's edge Phragmites australis dominated, while further along the slope herbs and grasses coexisted. In CDBs, the vegetation mostly consisted of grasses (e.g. Lolium perenne) with some herbs.

2.2 Experimental design

Throughout the polder, we selected 20 ditch borders (DBs) of which in 16 DBs the topsoil was removed to create a RDB and four DBs which were left intact (CDB). We included more regenerative borders to account for the variation in slopes and management after excavation. In each of the selected DBs we established four sampling points along a linear transect starting at the water's edge and going perpendicular to the ditch into the agricultural field, at 40 cm, 80 cm, 360 cm and 640 cm from the water's edge. Given that the RDBs were typically five meters wide, at these ditches, the distances 40 cm, 80 cm

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and 360 cm were located in the excavated area and distance 640 cm was located in the agricultural field.

In all DBs a permanent datalogger (TOMST, TMS-4) was installed at distance 360 cm to record soil moisture content (at a depth of -14 cm; Time-Domain transmission method using 2.5 GHz electromagnetic pulses) and soil and air temperature (at soil depth of -6 cm, and +2 cm and +15 cm above the soil surface; MAXIM/DALLAS/ semiconductor DS7505U+ sensors), measured every 15 minutes (Wild et al., 2019). The dataloggers were protected from disturbance by livestock with stainless steel bars and were running throughout the experiment. Data was extracted using Lolly software (version 1.51). For every DB, we characterized the height profile of the study area by measuring the height relative to the ditch water level at 40 cm intervals perpendicular to the ditch (0 cm) to the fourth distance (640 cm) using a theodolite in April 2023. The height profile was made at a 1 m distance from the measurement transect to minimize disturbance. We also measured the height at each transect point relative to the ditch water level once.

2.3 Soil respiration measurements

Soil respiration (R_{soil}) was measured five times throughout the period between June 2023 and March 2024 (June-August-October-January-February/March) at each distance at the water's edge across all DBs. Additionally, we measured soil respiration more frequently at a subset of the DBs (2 CDBs and 4 RDBs in July-September-December) to ensure we captured adequate seasonal variation in soil respiration. CO_2 was measured using a canopy assimilation chamber (145 mm × ⌀ 146 mm) connected to an infrared gas analyzer (IRGA; EGM-5 Portable CO_2 Gas Analyzer, PP Systems), equipped with a soil moisture (mV) and temperature probe (°C) measuring at a soil depth between 0 and -5 cm (Stevens HydraProbe). The chamber was equipped with a PAR sensor and a temperature sensor (Precision Thermistor). A plastic skirt was attached to the chamber over which a steel chain was placed to prevent air leakage during measurements. The chamber was covered with a light-blocking cover. Prior to the measurements, all the above-ground vegetation in the measurement area was clipped. To avoid excessive compression/disturbance of the soil close to the chamber, a ladder was placed on the

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adjacent soil from which the measurements were executed. The chamber CO_2 concentration (in ppm), soil moisture content, soil and air temperature, and PAR were measured every second over a period of two minutes of which the first 12 seconds were discarded.

The CO_2 flux rate (F_{rate}) for each sampling moment was then calculated (see eq. 1) (Möhl et al., 2023):

$$F_{rate} = \frac{\Delta\, CO_2 \times P \times 273.15}{22.4 \times (273.15 + T_{air}) \times P_i} \times \frac{V}{A} \times 1000 \ \ (\mu mol\ CO_2\ m^{-2}\ s^{-1}) \quad \text{(eq. 1)}$$

Where $\Delta\, CO_2$ is the estimated slope from a linear regression of CO_2 over time (Δ[CO2]/Δt), P the average air pressure during the measurement (kPa), 273.15 (K) the standard temperature, 22.4 (L mol^{-1}) the molar volume of CO_2, T_{air} the air temperature in the chamber (measured by the Precision Thermistor), P_i the average air pressure at sea level (101.325 kPa), V the volume (0.002427 m^3) of the chamber and A the area of the chamber cross section (0.0167 m^2).

The calculated F_{rate} was then converted to CO_2 (g) per m^2 per h using the molar mass of CO_2 (44.01 g mol^{-1}). Throughout the year, some air temperature measurements failed. To calculate the CO_2 flux in these cases, we performed a linear regression with air temperature measurements from the HydraProbe and from the permanent datalogger (+15 cm) to estimate the missing values for air temperature. We calibrated the HydraProbe, by taking a soil sample (14 cm × ⌀ 18.5 cm) which was cut out of the soil from one of the sites, watering it back to water holding capacity and then oven drying (30°C) the sample for a period of two weeks. Soil moisture values from the HydraProbe were recalculated based on the linear relationship between the HydraProbe values and gravimetric soil moisture content measured during this drying period.

We computed an annual dataset with soil temperature and soil moisture data collected with the IRGA during CO_2 flux measurements (5 or 8 measuring points). To enable an annual (one full year) interpolation between the measuring points we duplicated our dataset and incremented the year in the duplicated set, while leaving all other values

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unchanged. We then interpolated daily values for soil temperature and moisture by applying a smoother (span = 0.75) for the period between October 2023 – October 2024 (Fig. S7 and S8 in the Supplement).

2.4 Peat descriptions and soil core profiles

Morphological and structural descriptions were made of the peat soil at all DBs at a distance of 360 cm from the water's edge. We used a gouge auger (Eijkelkamp 04.02.SA; ⌀ 3 cm) to drill to the Pleistocene deposits (100 – 365 cm deep). All visibly distinct peat layers were identified and described following the description method by Deltares (Erkens et al., 2013).

We collected soil cores (100 cm x ⌀ 6 cm) at all DBs at a distance of 360 cm from the water's edge using a gouge auger (Eijkelkamp 04.02.SA). After sampling, we carefully sliced the cores out of the gouge using a steel wire and wrapped them in foil. The cores were then placed in PVC tubes and were wrapped in tubular plastic foil to prevent oxidation by air exposure. The cores were processed in the laboratory within 24 h after sampling. We took subsamples of the core at 5 cm depth intervals using a cutter (1 cm × 1 cm × 5 cm) to determine gravimetric moisture content (SMC) ($\frac{(SMC_{wet} - SMC_{dry})}{SMC_{wet}} * 100\%$; after drying at 70°C for 48 h), bulk density ($\frac{Dry\ weight}{volume}$) and organic matter content through loss on ignition (550°C for 4 h, muffle furnace). We determined total carbon (C) and nitrogen (N) content through flash combustion using an elemental analyzer after grinding the dried soil (FLASH EA 1112). For each of the 20 cores, carbon densities (kg C m^{-2}; Fig. S1 in the Supplement) were calculated for every 5 cm depth interval to 100 cm depth using the measured total C and bulk density values.

2.5 Groundwater levels and exposed carbon calculation

Groundwater levels were measured 11 times throughout the measuring period at a fixed site in the polder at 3 m distance from the ditch with a known height relative to the ditch water level. Groundwater levels were measured in four monitoring wells with different filter depths (at -25 cm, -50 cm, -100 cm and -150 cm). Due to high hydraulic head and

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311 semi enclosed aquifer, the groundwater level was above the phreatic groundwater level
312 in monitoring wells -100 cm and -150 cm and therefore the groundwater levels measured
313 in these wells were not used. We used the groundwater data from monitoring well -50 cm
314 for our calculations. To generate a daily time series of estimated groundwater level for
315 subsequent modelling, we applied a smoother (span = 0.4) to the data and extracted the
316 resulting interpolated daily groundwater level values. As the groundwater level was
317 measured at one site in the polder, we recalculated groundwater levels for all distances
318 to the water's edge in all DBs by performing a correction based on the relative height to
319 the ditch water level at the monitoring well to the relative height of all the distances to the
320 water's edge. We used these distance-specific interpolated daily ground water levels to
321 calculate daily exposed carbon which was defined as the sum of carbon densities in the
322 depth intervals above the interpolated groundwater level (kg C m^{-2}) (Fig. 2A, Fig. S9 and
323 S10 in the Supplement).

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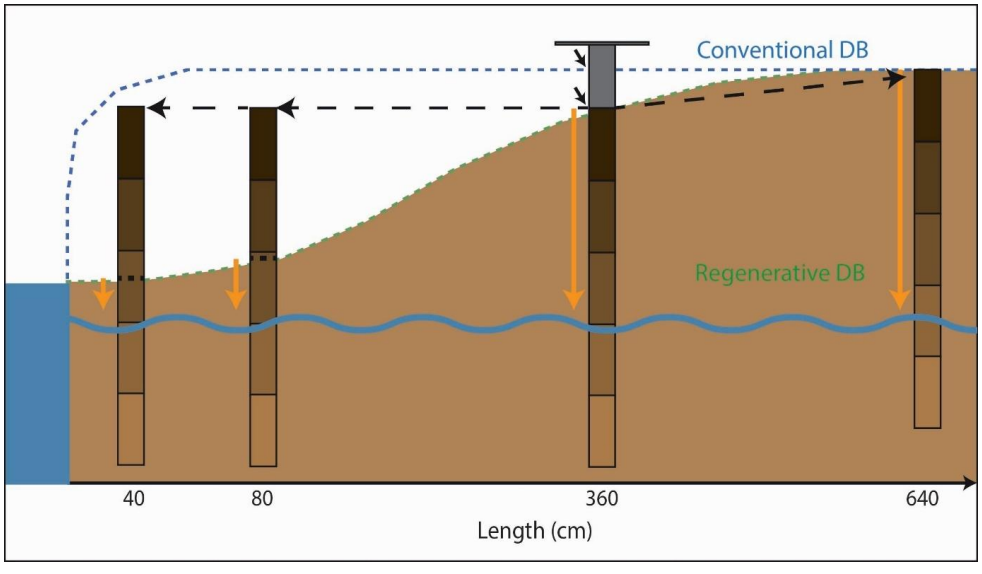

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326 *Fig. 2: Schematic representation of the exposed carbon calculation. We sampled 1 m*
327 *cores at distance 360 cm at all DBs (2 black arrows). In regard to the conventional DBs*
328 *(blue dashed line; CDB), we assumed that carbon content profiles were equal at all*
329 *distances to the water's edge and extrapolated the carbon densities at distance 360 cm*

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to all other distances to the water's edge. For the regenerative DBs (green dashed line; RDB), we also extrapolated the carbon densities at distance 360 cm to all other distances to the water's edges (black dashed lines) and for distances 40 cm and 80 cm we additionally corrected the carbon density profiles for TSR. TSR correction was performed based on their height differences and the start of the carbon density profile was adjusted based on this difference. Then exposed carbon values (orange arrows) were calculated based on the amount of carbon above the groundwater level (blue waved line).

2.6 Soil respiration modelling

To assess how soil temperature, soil moisture content and exposed carbon relate to spatiotemporal variation in soil respiration and to extrapolate our periodic field measurements to a full year of soil respiration under a variety of conditions we fit a series of statistical models to the respiration data. We compared model performance across four different parameterizations. Soil respiration (R_{soil}, F_{rate} (g CO2 m^{-2} h^{-1})), was modelled as the product of functions of temperature (T), moisture content (M) and exposed carbon (CE) (see eq. 2).

$$R_{soil} = R_{soil-T} * R_{soil-M} * R_{soil-CE} \quad \text{(eq. 2)}$$

For the R_{soil-T} and R_{soil-M} functions we chose model parameterizations that were identified as effective in previous soil respiration modelling studies (Suseela et al., 2012; Kechavarzi et al., 2010; Mäkiranta et al., 2009). Specifically, we assumed an exponential relationship between soil respiration and temperature.

$$R_{soil-T} = \mathrm{a}\, e^{(bT)} \quad \text{(eq. 3)}$$

Where R_{soil-T} is as defined in equation 2, a (>0) and b (0-1) parameters, and T the observed soil temperature (°C; soil depth up to -5 cm).

We assumed a parabolic relationship between soil respiration and soil moisture content. The moisture function we used requires fixed minimum and maximum values of soil

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moisture based on the data. We set the min and max to 10 and 100 % soil moisture content, respectively, to derive a parabolic function that predicted non-negative soil respiration over the full range of interpolated soil moisture data in our time series.

$$R_{soil-M} = (SMC - min)(max - SMC)^c \quad \text{(eq. 4)}$$

Where R_{soil-M} is as defined in equation 2, SMC the observed soil moisture content (%), min a fixed minimum SMC (10), max a fixed maximum SMC (100) and c (>0) a parameter to be fit.

We tested models with three possible relations between soil respiration and exposed carbon describing a linear relationship with a fixed intercept at 0 (when Cexp = 0 = $R_{soil-CE}$), a linear relationship with a free intercept (when Cexp = 0, $R_{soil-CE}$ = h) and a sigmoidal relationship (see eq. 5, 6 and 7).

$$R_{soil-CE} = Cexp$$

$$R_{soil-CE} = h + iCexp$$

$$R_{soil-CE} = \frac{l}{1 + e^{(-n\,Cexp)}} \quad \text{(eq. 5, 6, and 7)}$$

Where $R_{soil-CE}$ is as defined in equation 2, Cexp the computed amount of carbon above the groundwater level on the day of the F_{rate} measurement (kg C m^{-2}), h (>0), i (>0), l (>0), and n (>0) parameters.

We tested one soil respiration model with R_{soil-T} and R_{soil-M} only and three models additionally including one of the three $R_{soil-CE}$ sub-models which led to four fitted soil respiration models:

$$R_{soil} = \left(a\,e^{(bT)}\right)((SMC - min)(max - SMC)^c)$$

$$R_{soil} = \left(\mathrm{d}\,e^{(fT)}\right) * ((SMC - min)(max - SMC)^g) * Cexp$$

$$R_{soil} = \left((h + iCexp)\,e^{(jT)}\right) * ((SMC - min)(max - SMC)^k)$$

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$$R_{soil} = \left(\left(\frac{l}{1 + e^{(-n\,Cexp)}} \right) e^{(mT)} \right) * ((SMC - min)(max - SMC)^{o})$$

(model 1, 2, 3, and 4, respectively)

All four soil respiration models were fit using Bayesian non-linear regression in the 'brms' package (Bürkner, 2017). For all models, four Markov chain Monte Carlo (MCMC) chains were ran with 2000 iterations each and with a 1000 iteration warm-up period. MCMC chain convergence was evaluated diagnostically through trace plots and density plots. The four models were compared with Leave-One-Out Cross-Validation (LOO-CV), comparing the four models by their computed expected log pointwise predictive density (elpd_diff) with uncertainty (se_diff), expressed relative to the top-ranked model (which has elpd_diff = 0). We also evaluated the posterior predictive accuracy of the four models by computing the mean squared error (MSE) by generating posterior-predictive draws and calculating the mean squared error between the predicted and observed soil respiration values (mse). Based on these two model evaluation methods, we chose the best performing model and evaluated the model fit by a residual vs. fitted plot and an observed vs. predicted plot (Fig. S3 and S4 in the Supplement).

2.7 Carbon debt and payback time: assumptions and calculation method

We used the model estimates to estimate the 'payback time' defined as the number of years for net negative effects on cumulative CO_2 emissions (net reduced emissions), to be reached after implementing RDBs. The payback time indicated how many would be necessary to compensate for soil carbon lost as a result of the intervention. The payback time was computed by dividing the amount of carbon removed by TSR (C-removed) by the difference in soil respiration between CDBs and RDBs (ΔR_{soil}). Our approach was based on three simplifying assumptions: 1) all organic carbon removed during the TSR procedure was mineralized to CO_2 i.e. the carbon debt was equivalent to the amount of soil carbon in the removed soil volume; 2) no difference in net primary production between CDBs and RDBs and, thus, relative changes in carbon stock between border types were solely driven by differences in soil respiration rates; and 3) the differences in

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413 conditions driving R_{soil} (temperature, moisture content and exposed carbon) remained
414 constant over time (Fig. 3a).

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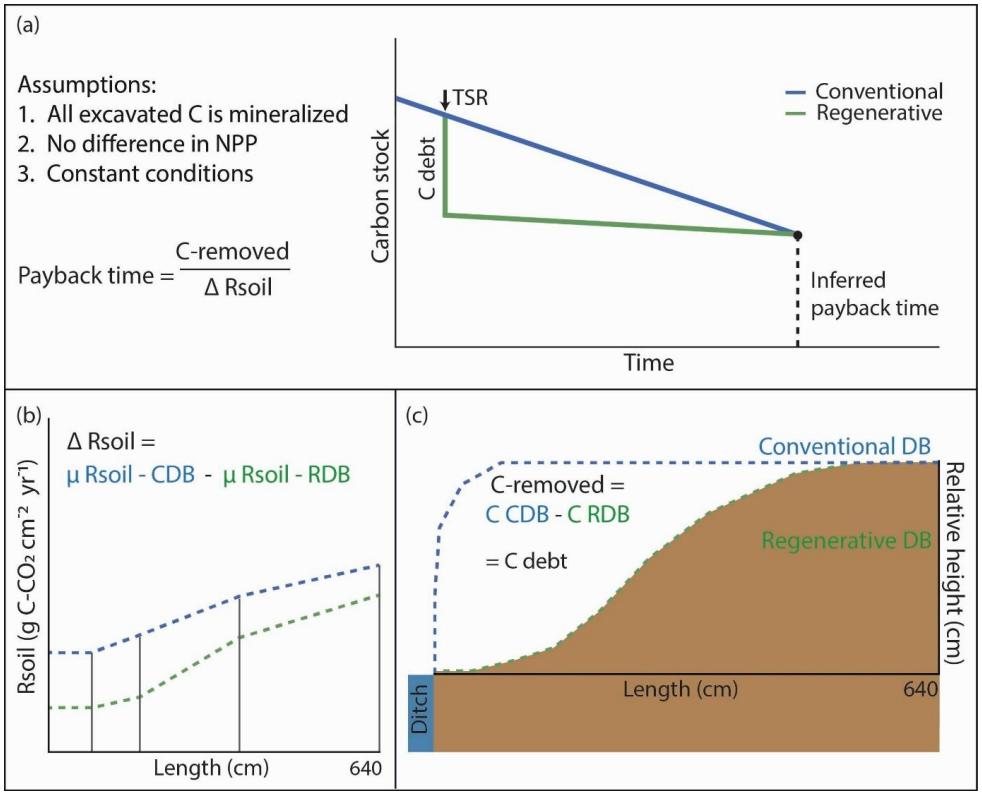

416

417 *Fig. 3: Payback time calculation. (a) The calculation of the inferred payback time was*
418 *based on three assumptions: (1) all excavated carbon (C) was mineralized, (2) no*
419 *difference in net primary production (NPP) between border types and (3) constant*
420 *conditions over time. The assumptions imply that the difference in carbon stock between*
421 *conventional (CDBs; blue) and regenerative ditch borders (RDBs; green) was solely driven*
422 *by their different soil respiration. (b) Δ R_{soil} was computed by subtracting the mean (μ) $R_{soil-CDB}$*
423 *from the μ $R_{soil-RDB}$ over the entire length of the transect (0 - 640 cm). C. C-removed*
424 *(carbon debt) was computed by subtracting the mass of carbon above the ditch water*
425 *level RDBs from the corresponding carbon stock of CDBs.*

426

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As our soil respiration measurements only started at 40 cm from the water's edge, we also had to assume the area in between 0 and 40 cm showed the same respiration rates and dynamics as the 40 cm measurements and therefore an additional distance of 0 cm was added to the dataset with data identical to the 40 cm data (Fig. 3b). To determine Δ R_{soil}, annual soil respiration was calculated at every distance to the water's edge (g CO2 cm^{-2} yr^{-1}) using the 'trapz' function (R package pracma). Distance to the water's edge was plotted as an actual distance using a smoother (span = 0.5) between 0-40-80-360-640 cm of which the data was extracted (Fig. 3b). With this data, we calculated the mean annual soil respiration rate for both DB types by numerical integration using 'trapz'. Δ R_{soil} (g C-CO_2 cm^{-2} yr^{-1}) was then computed by subtracting the mean rate of CDBs with the mean rate of RDBs (Fig. 3b). To determine C-removed, we used 'trapz' to calculate the total volume of soil under the relative height profiles (volume above water level in ditch reaching till distance 640 cm into the field) of each DB per meter of ditch border length. The average of the total volume of all CDBs functioned as a pre-TSR reference of which the total volume of each RDB was subtracted which was defined as 'removed volume' (cm^2) (Fig. 3c). We calculated the mean carbon density in three ways: (method 1) a mean carbon density for each regenerative DB; (method 2) a mean carbon density of the four conventional borders; and (method 3) a mean carbon density of all the cores combined. The three mean carbon densities were then multiplied with the 'removed volume' of each RDB which was defined as C-removed (g C cm^{-2}).

Mean payback time was computed by dividing the generated values of C-removed by the generated values of Δ R_{soil}. We calculated associated uncertainty by performing parametric bootstrap simulations (Efron & Tibshirani, 1994). We generated a 1000 draws from normal distributions of C-removed (using the standard error derived from carbon densities) and of Δ R_{soil} (calculated with three methods using the standard error derived from difference between the mean rate of conventional borders and regenerative borders).

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2.8 Statistical analysis

We extracted the parameter estimates of the best performing model (model 3, see Results) and used them to calculate daily modelled soil respiration rate values (g CO_2 m^{-2} h^{-1}) with the interpolated daily temperature, soil moisture content and exposed carbon values for each distance to the water's edge (example in Fig. S2 in the Supplement). These daily modelled values were then used to compute estimated annual soil respiration values for each distance to the water's edge. Due to missing soil temperature and soil moisture content observations, we excluded distance 40 for DB numbers 1C, 8C and 12 from further analyses. To analyze the differences in potential drivers across management types and within DB, we fit separate linear mixed-effects models to the mean soil temperature, mean soil moisture content, mean exposed carbon, and annual modelled soil respiration for each distance to the water's edge with DB type (categorical) and distance to the water's edge (categorical) as fixed effects and DB number as random effect (Bates et al., 2015). We tested for normality of the residuals using Q-Q plots and for homogeneity of the variance using residual vs. fitted plots. The models included both main effects and the two-way interaction and significance of terms was tested using ANOVA (Kuznetsova et al., 2017). We performed planned contrasts to test for differences between DB types averaged over each distance to the water's edge and between DB types within each distance to the water's edge ($\alpha = 0.05$) (Searle et al., 2012). The same analysis, for annual modelled soil respiration, was repeated without distance 640 cm, as this distance was located in the agricultural field where differences between DB types were reasoned to be minimal. Excluding distance 640 cm did not affect the conclusions of the ANOVA and planned contrast analyses and the distance was kept for analysis.

To quantify the relative importance of each of the three predictor variables in contributing to the model predictions, we computed Shapley Additive Explanations (SHAP) using the 'fastshap' package (Štrumbelj & Kononenko, 2013). For each SHAP evaluation, the posterior expected value was computed by averaging 200 draws from the posterior predictive distribution. SHAP values were estimated with Monte Carlo approximation using 100 simulations per observation, using the full randomized version of the observed dataset. These values represented each predictor's contribution to the model

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predictions by averaging its marginal impact across all possible subsets of the remaining predictors.

We performed counterfactual decomposition to determine the relative contributions of the predictors soil temperature, soil moisture content and exposed carbon in explaining differences in soil respiration between DB types (Fairlie, 2005). To investigate seasonal and spatial variation in variable importance, we performed this procedure using model predictions at all distances to the water's edge and over the entire measuring period. The predictor values were first averaged over the DB numbers for each DB type × distance to water's edge × date combination to obtain the daily contributions of the predictors. Soil respiration predictions were made using model 3 and its parameter estimates (i.e. h, i, j, and k). Soil respiration at CDBs was used as the reference condition which meant the values of the predictors soil temperature, soil moisture content and exposed carbon correspond to soil respiration at CDBs for each distance to the water's edge × date. We then generated counterfactual predictions in which the value of each predictor, one predictor at once, was replaced with the corresponding predictor values at RDBs, while holding the remaining two predictor CDB values. This was repeated for all three predictors, and over the entire measuring period.

3 Results

3.1 Patterns of soil respiration and soil descriptions

Soil temperature measurements made during respiration measurement campaigns displayed seasonal variation from 33.7 °C summer to -0.3 °C in winter, and corresponding respiration measurements showed an overall positive relationship to soil temperature (Fig. 4a). The relationship between soil moisture content and soil respiration was less clear for both DB types (Fig. 4b). Measured soil moisture content ranged from 30 % to 83 %. Overall, soil respiration increased with increasing exposed carbon. Exposed carbon ranged from 0 kg C m^{-2} in winter when groundwater levels were at the soil surface up to 171 kg C m^{-2} in summer when groundwater levels were low (Fig. 4c). Peat depth at the 20 DBs varied between 100 cm (DB 1A) and 329 cm (DB 6) (Fig. 4d). There was also

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514 considerable variation in the relative distribution and thickness of the different soil layer
515 types, in particular DB 13 was rich in clay, which could suggest a previous riverine history,
516 and DB 7 had quite a large, disturbed gravel layer in the top 1 m. The botanical origin of
517 the majority of the peat was identified as reed (*Phragmites australis*).

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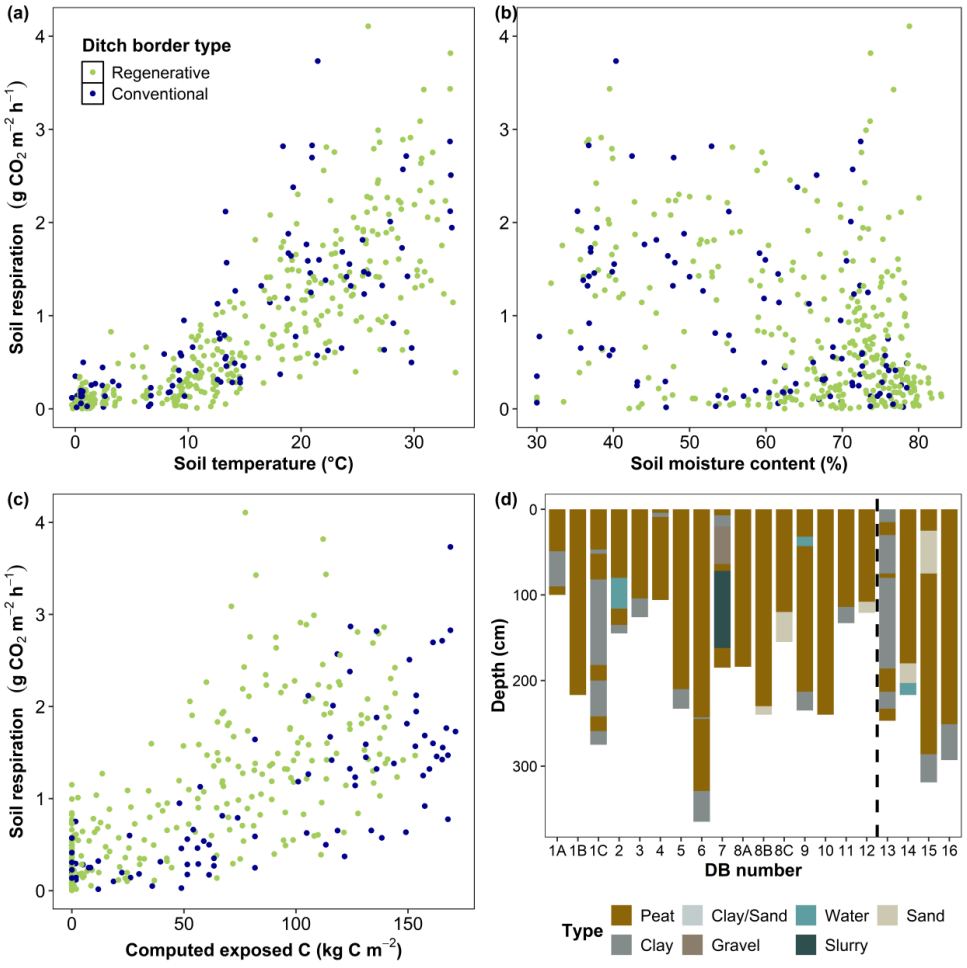

519

520 *Fig. 4: The relationship between measured soil respiration and (a) soil temperature, (b)*
521 *soil moisture content, and (c) computed exposed carbon. Points are colored by DB type*
522 *(regenerative = green (RDB) and conventional = blue (CDB)) and represent the observed*
523 *field data (n = 445). (d) Soil profiles (colors represent different layer types) of all DBs down*
524 *to the mineral layer; the dashed line separates the RDBs (n = 16) from the CDBs (n = 4).*

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3.2 Model fit evaluation & model predictions

The LOO-CV results indicated model 3 to be the best of the candidate models for explaining the observed variation in soil respiration (Table 1). The same conclusion was drawn based on comparing the MSE, where model 3 had the lowest MSE. Model 3 was therefore selected as the best performing model and used for subsequent calculations.

Table 1: Model comparison results of the Leave-One-Out Cross-Validation (LOO-CV) and Mean Squared Error (MSE) for the four soil respiration models with four possible different exposed carbon sub-models (see section 4.6). Using LOO-CV, we computed and compared the log pointwise predictive density with uncertainty (elpd_diff ± se_diff) of each model, expressed relative to the best ranked model (elpd_diff ± se_diff = 0). Using MSE, we compared the posterior predictive accuracy of the models by computing the mean squared error between the predicted and observed soil respiration values (mse). The resulting best performing model, resulting from the two model evaluation methods, is highlighted in bold.

Models	Exposed C sub-model	Elpd_diff ± se_diff	Mean squared error
1.	-	-28.5 ± 10.2	0.543
2.	~ $Cexp$	-47 ± 9	0.585
3.	**~ h+$iCexp$**	**0 ± 0**	**0.472**
4.	~ $\frac{l}{1+e^{(-n\,Cexp)}}$	-3.5 ± 6.6	0.48

Based on the computed SHAP values, the predictor variables exposed carbon and soil temperature had the highest contributions to explaining overall variation in soil respiration (model 3), explaining 0.321 and 0.303 of the total variance, respectively (Fig. S5 in the Supplement). Soil moisture content explained 0.08. of the variance and together the three predictor variables explained 0.704 of the total variance (70 %).

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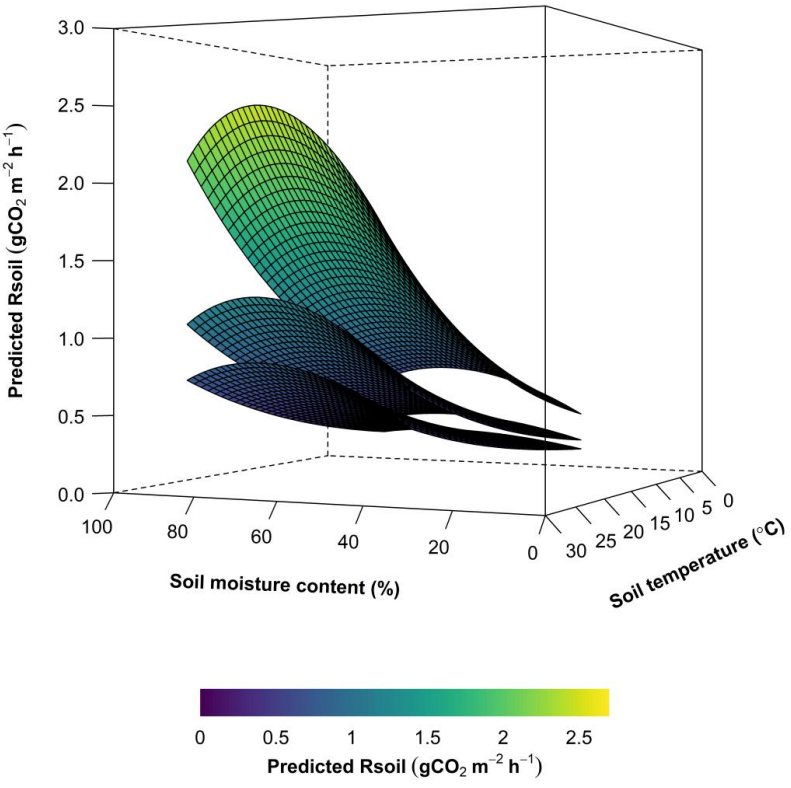

548

549 *Fig. 5: Predicted soil respiration (R_{soil}; g CO_2 m^{-2} h^{-1}) for soil temperature (°C), soil moisture*
550 *content (%) and three levels of exposed carbon (kg C m^{-2}). Soil respiration predictions*
551 *were generated based on model 3 (h = 5.2e-4 ± 1.2e-4; i = 7.7e-6 ± 2.7e-6; j = 4.4e-2 ±*
552 *0.41e-2; and k = 0.6 ± 0.071) using the observed range of soil temperature and soil*
553 *moisture content. Separate prediction surfaces were plotted for the 10th, 50th and 90th*
554 *percentile of exposed carbon (10th = 0; 50th = 34.2; and 90th = 134.3 kg C m^{-2}). Positions*
555 *of the surface relative to the vertical axis and the colour gradient indicative values of*
556 *predicted soil respiration.*

557

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Soil respiration increased with increasing amounts of exposed carbon (Fig. 5 & Fig. S6 in the Supplement). For example, at the annual mean soil temperature of 17 °C and soil moisture content of 56 %, soil respiration increased by 0.367 g CO_2 m^{-2} h^{-1} when exposed carbon increased by 1 SD. Soil respiration rates peaked at a 65 % soil moisture content (Fig. 5).

3.3 Importance of soil temperature, soil moisture content and exposed carbon varied over DB type and distance to the water's edge

DB type and distance to the water's edge had no significant effect on mean annual soil temperatures (Table 2, Fig. S7 in the Supplement). DB type significantly affected the mean soil moisture content. RDBs soils were 11 % wetter than the CDBs when averaged over the entire measurement period, with RDBs having a mean of 66 ± 0.067 % (SE) and CDBs of 59 ± 0.15 % (Table 2, Fig. S8 in the Supplement). Along the distances, 40 cm and 80 cm were 14 % and 15 % wetter than 360 cm and 22 % and 23 % than 640 cm, respectively. DB type significantly affected the amount of exposed carbon at all the distances to the water's edge (Table 2, Fig. S9 in the Supplement). RDBs had 40 %, 48 %, 49 %, and 38 % lower exposed carbon than CDBs, respectively from the water into the field.

Table 2: ANOVA table of the linear mixed-effects model with the observed data comprising of mean soil temperature, mean soil moisture content, mean computed exposed carbon, and with modelled predictions comprising of annual modelled soil respiration as response variable; ditch border type (conventional and regenerative) and distance to the water's edge (40 cm, 80 cm, 360 cm and 640 cm) as fixed effects; and ditch border number (1-20) as random effect. The significant p values are highlighted in bold.

	Ditch border type	Distance to the water's edge	Ditch border type × Distance to the water's edge

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		F_1	*p*	F_3	*p*	F_3	*p*
Observed data (n = 80)	Mean soil temperature	0.6	0.467	0.8	0.478	1.4	0.26
	Mean soil moisture content	8.4	**0.01**	24.5	**<0.001**	2	0.123
	Mean computed exposed carbon	32.2	**<0.001**	27.1	**<0.001**	2.9	**0.041**
Modelled predictions (n = 77)	Annual modelled soil respiration	9.5	**0.006**	9.1	**<0.001**	0.4	0.749

The computed seasonal patterns in soil temperature, moisture and exposed carbon and the model fit together determined the seasonal patterns in predicted soil respiration (Fig. 6). Along the distances, the mean predicted soil respiration rate was 0.24 ± 2e-3 t CO_2 ha^{-1} d^{-1} in CDBs, exceeding the 0.20 ± 0.88e-3 t CO_2 ha^{-1} d^{-1} predicted in RDBs. During spring and summer, the difference in soil respiration between CDBs and RDBs was most apparent at distances 40 and 80 cm (Fig. 6). Where at 40 and 80 cm RDBs emitted on average 0.28 ± 2.2e-3 and 0.29 ± 2.2e-3 t CO_2 ha^{-1} d^{-1} and CDBs 0.35 ± 4.6e-3 and 0.37 ± 4.8e-3 t CO_2 ha^{-1} d^{-1}, while at 360 and 640 cm RDBs emitted on average 0.33 ± 2.1e-3 and 0.33 ± 2e-3 t CO_2 ha^{-1} d^{-1} and CDBs 0.37 ± 3.5e-3 and 0.36 ± 3.8e-3 t CO_2 ha^{-1} d^{-1} (respectively).

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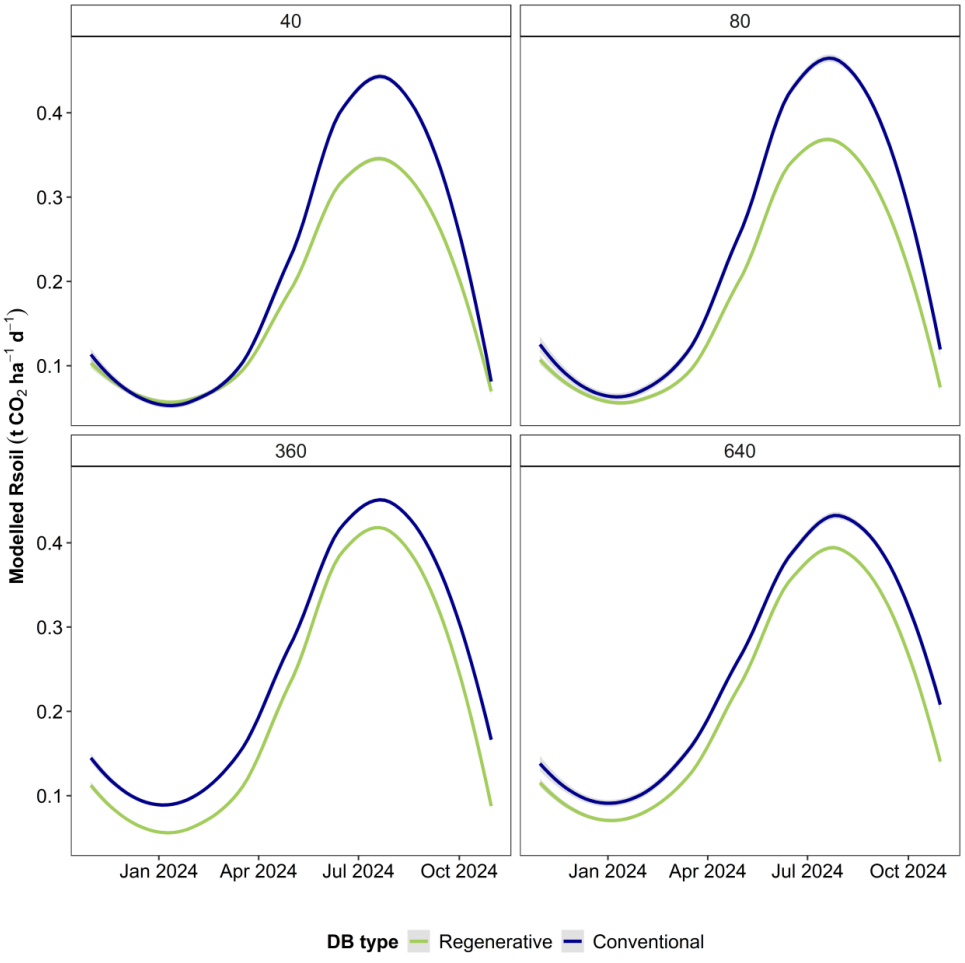

595

596 *Fig. 6: Modelled soil respiration (R_{soil}; t CO_2 ha^{-1} d^{-1}) over a one-year period showing the*
597 *mean effect of regenerative (green; RDB) and conventional (blue; CDB) DBs at the four*
598 *distances to the water's edge (40 cm, 80 cm, 360 cm and 640 cm). Model predictions*
599 *were computed from the annual interpolated datasets of soil temperature, soil moisture*
600 *content, and exposed carbon using the model parameter estimates from model 3. The*
601 *fitted lines represent the overall modelled soil respiration for each DB type (loess*
602 *smoothed conditional means, span = 0.75).*

603

604 The relative importance of the three different predictor variables in explaining variation
605 between DB types depended on the distance to the water's edge. At distance 40 and 80

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cm, there was almost no difference in soil respiration between CDBs and RDBs (Fig. 7). During spring and summer the difference between DB types was more profound (Fig. 6) which was driven by differences in exposed carbon (Fig. 7). Lower amounts of exposed carbon in RDBs resulted in lower respiration rates in the spring and summer period (Fig. 6, Fig. S9 in the Supplement). The contributions of variation in soil temperature and soil moisture content remained near zero throughout the entire measuring season at these distances. In contrast, at distances 360 and 640 cm, predicted soil respiration was slightly lower in RDBs throughout the entire measuring season due to lower amounts of exposed carbon and this effect was more pronounced during spring and summer (Fig. 7). Also for these distances, lower amounts of exposed carbon in RDBs resulted in lower respiration rates throughout the year and most strongly in spring and summer (Fig. 6 & Fig. 7, and Fig. S9 in the Supplement). The contributions of variation in soil temperature and soil moisture content were minimal during winter, however became more pronounced during spring and summer (Fig. 7). Higher temperatures and soil moisture content during this period resulted in higher respiration rates in RDBs (Fig. 6, Fig. S7 and S8 in the Supplement). The opposing contributions of exposed carbon on the one hand and soil temperature and moisture content on the other hand therefore diminished any substantial differences between the DB types at 360 and 640 cm. However, averaged over the whole year and across the four distances to the water's edge, the overall higher contribution of exposed carbon resulted in lower soil respiration in RDBs throughout the year (Fig. 7).

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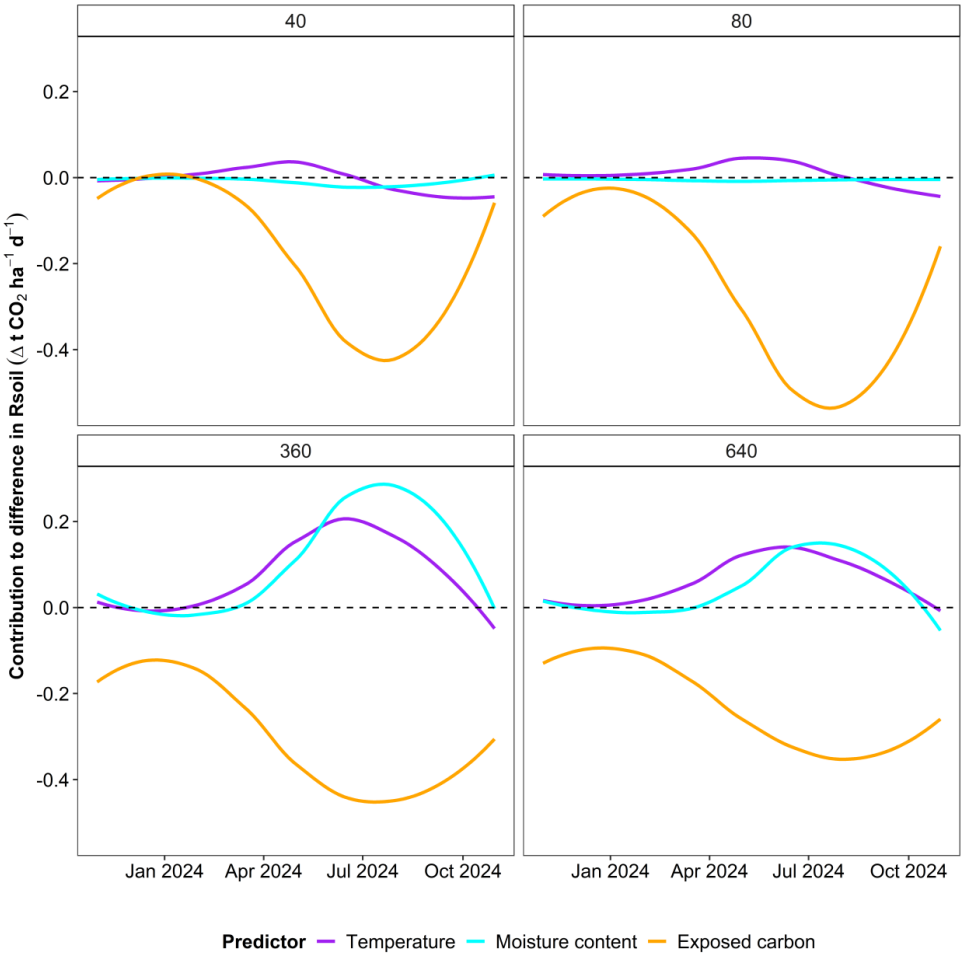

Fig. 7: Contributions of variation in the predictors soil temperature (purple), soil moisture content (blue) and exposed carbon (orange) to differences in modelled soil respiration (Δ g CO_2 ha^{-1} d^{-1}) between conventional (CDB) and regenerative (RDB) DBs for the four distances to the water's edge (40 cm, 80 cm, 360 cm and 640 cm). The contributions were computed with counterfactual decomposition. Lines represent the mean daily contribution of each predictor to the difference between CDBs and RDBs with CDB as the reference. Positive values indicate that the difference in the predictor variable between the DB types at the corresponding time point leads to higher predicted respiration rates and negative values indicate that the difference in the predictor variable between the DB types at the corresponding time point leads to lower predicted respiration rates in

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regenerative DBs. A value of 0 indicates that there is no difference in the predictor variable between the DB types and therefore does not contribute to a difference in predicted respiration rates. The sum of the daily contributions of soil temperature, soil moisture content, and exposed carbon is equal to the total difference in mean modelled soil respiration between CDBs and RDBs at the corresponding time point (visible in Fig. 6).

3.4 Regenerative ditch borders had lower predicted soil CO_2 emissions

Modelled total annual soil respiration differed significantly between DB types (Table 2). Averaged across the four distances, RDBs emitted an estimated 19.9 ± 0.46 t CO_2-C ha^{-1} yr^{-1} and CDBs 24.1 ± 0.6 t CO_2-C ha^{-1} yr^{-1}, a reduction of 17 % (Fig. 8). Distance to the water's edge also affected the annual modelled soil respiration (Table 2), where distance 40 cm emitted 18 ± 0.84 t CO_2-C ha^{-1} yr^{-1} which was 18 % less than the 22.2 ± 0.64 and 22.3 ± 0.78 t CO_2-C ha^{-1} yr^{-1} emitted at distances 360 and 640 cm, respectively.

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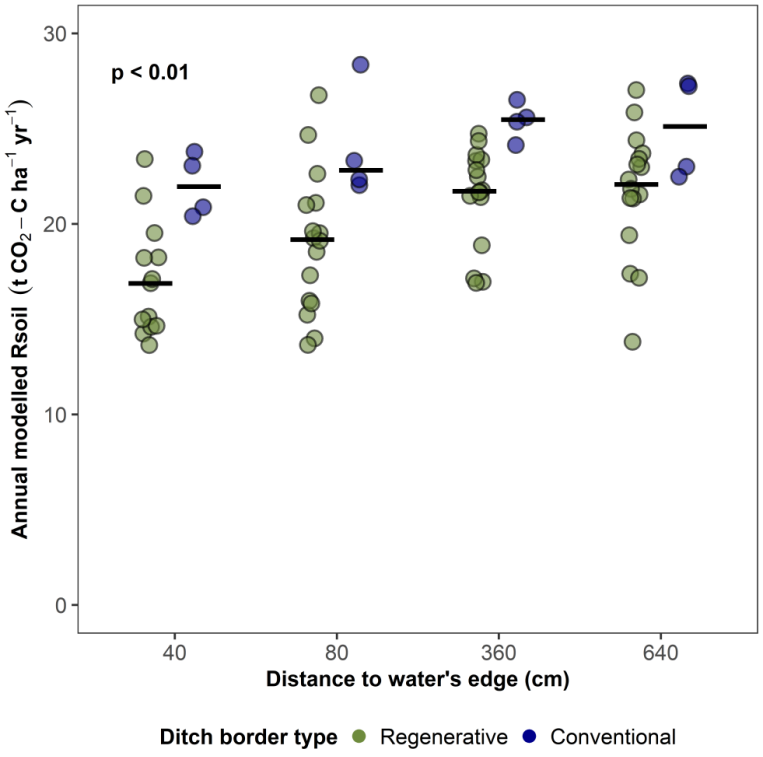

Fig. 8: Annual modelled soil respiration (R_{soil}; t CO_2-C ha^{-1} yr^{-1}) for the four distances to the water's edge (40 cm, 80 cm, 360 cm and 640 cm) in conventional (blue; CDB) and regenerative (green; RDB) DBs. Each point (jittered) was derived from predictions made based on the interpolated annual datasets of soil temperature, soil moisture content and exposed carbon using model estimates (model 3) for each DB × distance to the water's edge combination. Each black bar shows the mean annual modelled soil respiration for each DB type × distance to the water's edge combination. P-value represents the overall effect of DB type, computed from a linear mixed-effects model.

3.5 Carbon debt increased time before net reductions in CO_2 emissions can be reached

The estimated removed carbon due to TSR (carbon debt) varied depending on which method was used to quantify the carbon density and ranged between 450 ± 4.2 (method

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1) – 458 ± 0.6 (method 2), and – 490 ± 1.5 (method 3) g C cm^{-2}. The integrated mean respiration rate in CDBs was estimated at 0.248 ± 2.9e-3 g C cm^{-2} yr^{-1} and in RDBs at 0.207 ± 2.4e-3 g C cm^{-2} yr^{-1}. Combining these estimates, resulted in an estimated payback time of 109 ± 39 (method 1), 119 ± 42 (method 2), or 111 ± 40 (method 3) (parametric bootstrap, SE) years until RDBs show net reduced CO_2 emissions in comparison to CDBs.

4 Discussion

We showed that incorporating exposed carbon into models with soil temperature and soil moisture content led to improved predictions of soil respiration. Exposed carbon and soil temperature explained the majority of the variation in soil respiration. Soil respiration peaked at 65 % soil moisture content and was lower in RDBs in comparison to CDBs over the whole transect. This difference was greatest during spring and summer. While at distances 40 and 80 cm reductions in predicted soil respiration were driven by lower amounts of exposed carbon, at distances 360 and 640 cm raised soil temperatures and moisture content mostly cancelled out reductions in soil respiration by lower amounts of exposed carbon. Exposed carbon was the foremost driver in differences between CDBs and RDBs and soil temperature mostly drove temporal patterns in soil respiration. On an annual basis, RDBs reduced soil CO_2 emissions by 17 % in comparison to CDBs. However, additional calculations indicate that this reduction due to RDBs, and the potential carbon debt incurred if carbon in removed topsoil is fully mineralized, imply that a period of multiple decades is required before DB management results in a net reduction in cumulative CO_2 emissions relative to the status quo.

4.1 Soil respiration modelling: exposed carbon and soil temperature drive variation in soil respiration

The potential CO_2 production from aerobic peat topsoil in the Netherlands can reach 19.9 ± 13.6 t CO_2-C ha^{-1} yr^{-1} (i.e. 200 ± 137 kg CO_2 ha^{-1} d^{-1}) (Boonman et al., 2025). Our estimates of annual CO_2 productions are consistent with Boonman et al. (2025) which

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provides support for the model predictions and estimates reported in this study. Soil moisture content explained only a small proportion of variation in soil respiration, however showing consistent patterns with other studies. Our model predicted the highest soil CO_2 production at a moderate soil moisture content (65 %), which likely reflects the balance between opposing effects of soil moisture on O_2 availability and substrate diffusion (Fairbairn et al., 2023; Linn & Doran, 1984). Exposed carbon and soil temperature both explained a substantial part of the variation in soil respiration. Soil temperature is known to drive seasonal variation in soil respiration rates (Lloyd & Taylor, 1994; Minkkinen et al., 2007; Mäkiranta et al., 2009). The importance of exposed carbon in our soil respiration model indicated that the amount of material available to aerobic decomposition processes is also influencing soil respiration rates. A previous analysis of annual carbon budgets in different peatland sites across the Netherlands found that higher values of net ecosystem carbon balance (i.e. carbon loss from the soil-vegetation system) correlate with higher amounts of exposed carbon (Aben et al., 2024). In this study we showed that exposed carbon also drives temporal variation in soil respiration between peat meadows within one peat site.

Although soil temperature, soil moisture content and exposed carbon are known to be key drivers, there are other important (biotic) drivers of soil respiration that were not included in this study and could drive variation in soil respiration unexplained by the predictors in our model. Root exudates are easily decomposable compounds and can promote fast decomposition in soils (Kuzyakov, 2010). Vegetation shifts accompanying a change from CDBs to RDBs (i.e. from a *Lolium perenne* to *Phragmites australis* dominance) can result in changes in the supply of root exudates. *Lolium perenne* is known to release high amounts of root exudates into the soil which could have contributed to higher soil respiration rates at the conventional sites (Kuzyakov et al., 2001). Consequently, vegetation-induced effects could also possibly explain differences in soil respiration. This effect would be stronger close to the water's edge where almost all of *L. perenne* is replaced by *P. australis*. Besides vegetation, microbial communities can also respond to the induced changes in soil conditions. Changes in soil moisture conditions, for instance, can alter microbial, mesofaunal and macrofaunal communities

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and could therefore potentially affect soil respiration rates (Laiho et al., 2001; Mäkiranta et al., 2009; van Dijk et al., 2009; van der Laan et al., 2025).

It is important to note that our soil respiration rates could possibly be an overestimation of the actual soil respiration rates, due to some sources of disturbance. Firstly, the vegetation was clipped before every field measurement which is known to potentially overestimate measured soil respiration rates (Li & Sun, 2011). However, this effect would be equal for all measurements and is unlikely to affect our conclusions on the effects of DB management. Interpolation of soil respiration rates will have introduced additional uncertainty and possibly led to missing or overrating peaks of soil respiration. As the DB sites in this study were in close proximity to each other, it is likely that they experienced similar climatic conditions, and similar peaks which reduces the effect of dissimilarity between sites that were possibly uncaptured in our interpolated data. We therefore consider the effect of these potential sources of error on the validity of our overall results to be negligible.

4.2 Reduced soil respiration in regenerative ditch borders

RDBs effectively reduced soil respiration rates throughout the entire ditch border (17 %), which can be mostly attributed to lower amounts of exposed carbon throughout the year in comparison to CDBs. During prolonged periods of drainage, increased decomposition, consolidation and shrinkage coincide and compress and compact the aerated top layer (Galloway et al., 2016; Leifeld et al., 2011). These processes alter the soil properties and often result in a higher bulk density and mineral fraction, and a reduced pore volume in this layer. TSR removes the majority of this degraded amorphic topsoil and exposes a new less degraded peat layer (Van Den Berg et al., 2024). This is partly visible in our carbon density profiles, made at distance 360, which was more apparent at distances 40 and 80 cm due to deeper TSR depths here. The newly exposed topsoil layer is also known to contain lower levels of easily degradable carbon and to reduce the mobilization of dissolved organic carbon (Daun et al., 2026). This can all contribute to reduced soil CO_2 emissions (Harpenslager et al., 2015; Huth et al., 2020; Daun et al., 2026). However, reduced soil CO_2 emissions could be the result of changes in electron acceptor

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availability. RDBs had higher groundwater levels which control the oxygen availability in the peat soil and therefore aerobic peat decomposition which is captured in exposed carbon. It is likely that reduced exposed carbon through reduced oxygen availability is driving reductions in annual soil CO_2 emissions. Previous research on the effects of water infiltration systems to reduce peat decomposition also emphasize that the effectiveness of this measure is tightly linked to reductions in water table depth or exposed carbon (Aben et al., 2024). When these reductions were insufficient, the effect on the net ecosystem carbon balance was minimal.

So far, most studies have focused on TSR in combination with rewetting through groundwater management (Harpenslager et al., 2015; Zak et al., 2017; Huth et al., 2020; Quadra et al., 2023; Käärmelahti et al., 2024; Van Den Berg et al., 2024). In the case of RDBs, no changes in groundwater management take place after the gradual slope is constructed by TSR. We showed that RDBs were effective in reducing soil CO_2 emissions, however there are certain combinations of predictor conditions where RDBs have higher predicted soil respiration which may warrant some caution in applying this measure. In the middle of the RDB, the soil moisture content was around the optimum level for soil respiration in spring and summer (60-65 %). Under conditions where the reductions in exposed carbon are minimal to counteract the effects of optimal moisture conditions (e.g. minimal excavation & high groundwater table), elevated soil CO_2 emissions might occur. A similar effect occurred with subsoil irrigation, where water inputs during dry summer months raised moisture levels in the upper 40 cm into the range that optimizes microbial decomposition, thereby increasing CO_2 emissions (Weideveld et al., 2021). Nevertheless, we showed that TSR implemented when constructing RDBs along ditches is effective in reducing soil CO_2 emissions along the entire DB length. Our study suggests that TSR without additional rewetting interventions in the form of RDBs, can therefore be implemented to reduce CO_2 emissions while also providing biodiversity benefits (Renou-Wilson et al., 2019).

Our study focused solely on soil respiration, a complete carbon budget is necessary to assess the full impact of this measure. We already know vegetation shifts (Fig. 1) are occurring as a result of RDB construction (Bethe et al., 2025, shift to *Phragmites australis* dominance), which could be expected to result in differences in net primary production

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(NPP). *Phragmites australis* is known for its high biomass and primary productivity (Brix et al., 2001), and shifts to reed dominance could therefore lead to more biomass available for decomposition, which could therefore reduce the suppressing effect of RDBs on soil CO_2 emissions. However, measuring the full CO_2 balance remains difficult with the presence of *P. australis* due to the need for very large flux chambers. Another caveat in estimating the effectiveness of RDBs, is information on CH_4 emissions. CH_4 emissions usually start playing a role under inundated conditions and are usually minimal in drained peatlands (Hendriks et al., 2007; Huang et al., 2021; Boonman et al., 2025). Even though RDBs promotes wetter conditions and thus possibly CH_4 emissions, TSR typically ensures low CH_4 emissions in combination with wetter conditions (Huth et al., 2020; Käärmelahti et al., 2024; Van Den Berg et al., 2024). *Phragmites australis* growth can promote CH_4 emissions by enhancing CH_4 transport through aerenchyma or reduce CH_4 emissions by increased oxygenation of the sediment (Boonman et al., 2023; Minke et al., 2016). Although the effect of DB management on the net ecosystem carbon balance would give many insights, measuring it remains labor-intensive and costly.

4.3 Compensation of soil carbon lost during TSR may require decades

During the implementation of RDBs a substantial amount of soil carbon was removed and replaced. Although we determined an overall 17 % reduction in soil respiration in RDBs, it will take over 100 years to realize net reductions in cumulative CO_2 emission if the removed carbon is assumed to be fully mineralized. According to our calculation, the time elapsed since the establishment of RDBs in our study system is only one quarter of the total time required to realize a net benefit in terms of soil carbon balance relative to CDBs (Fig. 9a). However, the uncertain validity of the assumptions used in our calculations means that this conclusion is provisional at best. Firstly, the range of environmental changes induced by the TSR and border construction may also lead to differences in net primary production (NPP) between DB types. If this is the case, the payback rate would be determined not just by the difference in soil respiration that we observed, but would be further modified by differences in annual carbon inputs (Fig. 9b; black arrows). Although we did not quantify this in our study, there is evidence that

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different crops (*Azolla filiculoides*, *Typha angustifolia*, and *Typha latifolia*) used in paludiculture under similar conditions to our study site show different patterns in gross primary productivity in comparison to the original grassland with *L. perenne* (Van Den Berg et al., 2024). As the vegetation differed substantially between DB types, it's likely that above- and/or belowground productivity is likewise not equal. If productivity is greatly reduced in RDBs in comparison to CDBs, there is a chance that the carbon debt is not compensated for by our observed differences in soil respiration, which could in turn lead to an overall long-term carbon cost. Secondly, uncertainty arises from assuming all excavated carbon is mineralized. The excavated material could have been transferred to an environment with conditions unfavourable for mineralization (e.g. under higher water tables or anaerobic conditions), leading to a relatively lower initial carbon debt and consequently shorter payback time (Fig. 9c). If the proportion of the mineralized carbon from the removed soil reduces with 10 %, the payback time reduces with 11-12 years. Lastly, additional uncertainty arises from changes in soil abiotic conditions over the years following RDB establishment which could result in a progressively smaller difference in respiration rate between CDBs and RDBs and eventually halt the payback (Fig. 9d, green dashed lines). This could occur when peat build-up in RDBs is as fast as peat mineralization in CDBs, where after a certain number of years the C stock of both DB types converge. The faster convergence occurs, the earlier the RDBs would potentially start to show little to no net carbon benefit.

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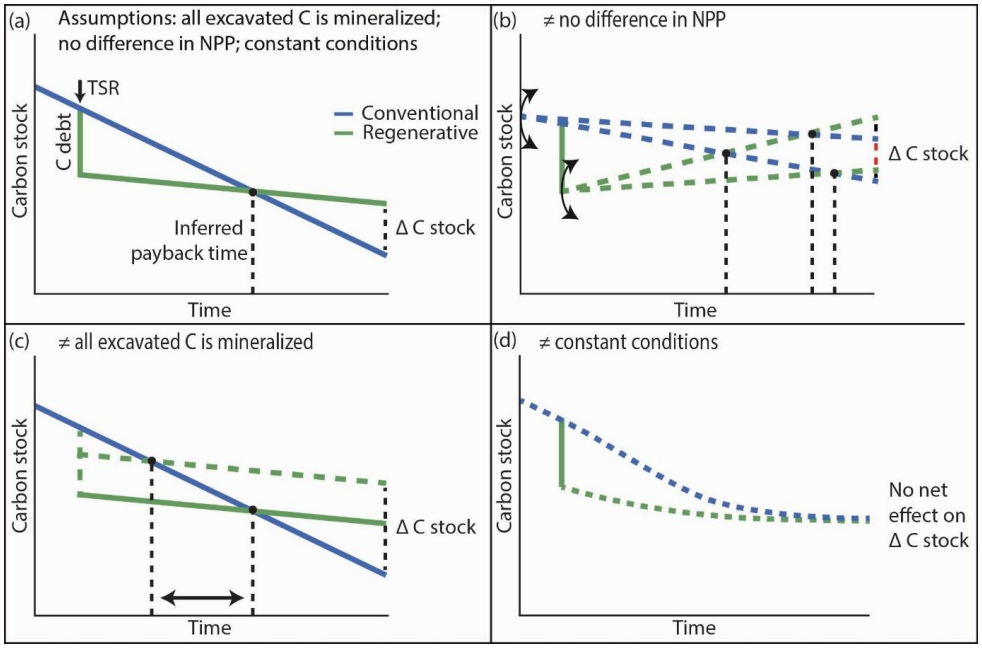

Fig. 9: (a) Conceptual figure of the carbon stock over time in relation to the resulting estimated carbon debt by TSR and the positive effect of TSR on the net carbon balance (Δ C stock) under three assumptions which then determine when reduced emissions were realized. The lines represent the observed difference in R_{soil} between regenerative (green; RDB) and conventional (blue; CDBs) DBs. (b) the possible effect on reduced emissions when NPP was not equal (assumption 2; black arrows). (c) the possible effects of the extent of the carbon debt (assumption 1; two green lines) on the resulting payback time. (d) the possible effects when the differences driving R_{soil} are not constant in time (assumption 3).

Similar studies looking at the carbon accounting consequences of land use conversion in peatland agroecosystems, reached broadly similar conclusions to our study. During a shift from intensive livestock farming to paludiculture, which is accompanied by partial removal of the topsoil, a reduction in CO_2 emissions is measured (Van Den Berg et al., 2024). However, it can take between 43-93 years (depending on the paludiculture species) for paludiculture sites to show reduced radiative forcing and start contributing to global warming reductions when incorporating emissions from the removed topsoil.

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This lag in realized positive effects is partly the result of emitted CO_2 from the removed topsoil which is equivalent to 27 years of emissions from the control conditions (Van Den Berg et al., 2024). Another study, estimates that reductions in radiative forcing start after 13-22 years (TSR depth of 30 cm) or after 40 years (TSR depth of at 60 cm) in comparison to an intensive grassland which is depending on the decomposition rate (conversion into CO_2) of the removed carbon (Huth et al., 2022). However, so far the exact fate of removed carbon after TSR and RDB implementation is not clear. Estimations of different fates and their associated mineralization rates, such as spreading over field, hole filling in field, and burying under anaerobic conditions, would greatly contribute to our estimations the timescale over which net emission reduction are achieved. This implies that any measures that lead to a minimalization of CO_2 emissions from the excavated carbon would further improve the peat polder-scale CO_2 emission reduction of RDBs with the use of TSR.

4.4 Management implications

Our study highlights the importance of creating RDBs surrounding grasslands used for livestock farming in Dutch peat meadows. TSR is often combined with groundwater table management (Huth et al., 2020; Quadra et al., 2023), however TSR used for RDBs (no rewetting) was effective in reducing soil CO_2 emissions. Reduced soil respiration was mainly obtained by reducing the relative amounts of carbon exposed to aerobic conditions. Implementation of RDBs creates a shift to a more regenerative farming system which would contribute to improved ecosystem services and biodiversity.

GHG emissions from ditches can also substantially contribute to the carbon budget in peatlands (25-33 % of the total landscape) (van der Knaap et al., 2025). So far, it's unclear whether implementing RDBs affects these emissions. RDBs have a higher perimeter-to-surface area ratio, enhanced soil-water contact, and therefore can possibly increase lateral transport of DOC, CO_2 and CH_4 to the ditch. Examining how RDBs affect these transport processes and the consequences for aquatic emissions would be needed to complete the effect of RDBs on the carbon balance.

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5 Conclusions

RDBs implemented with TSR, reduced soil CO_2 emissions which was predominantly driven by lower amounts of exposed carbon. Border areas around ditches could contribute to reduced CO_2 emissions, especially where restoration of entire fields is, at this moment, not possible due to economic and political motivations. In spring and summer, small hotspots for soil CO_2 production might occur when optimal moisture conditions are not counteracted by a sufficient reduction in exposed carbon. For this measure to be successful, it is crucial to process and store the excavated peat soil under conditions that limit mineralization. The predictors in this study (soil temperature, soil moisture content and exposed carbon) are relatively easy to monitor and, thus, allow easy monitoring of annual soil CO_2 production. The National Climate Agreement 2019 requires an emission reduction of 1 Mt CO_2-eq from Dutch peat meadows by 2030 (Rijskoverheid, 2019). This highlights the need for effective management measures, besides restoration where feasible. Regenerative measures, such as RDBs, could contribute to more responsible and peatland-adapted agriculture which is highly needed in wetland agroecosystems in northwest Europe (Freeman et al., 2022).

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Code availability

All code can be provided by the corresponding authors upon request.

Data availability

Data will be made available via Yoda.

Supplement

The supplement is available.

Author contributions

Sanne Bethe: Conceptualization, Methodology, Investigation, Formal analysis, Validation, Visualization, Writing – original draft, Writing – review & editing, Supervision.

Mariet Hefting: Conceptualization, Methodology, Writing – review & editing, Supervision.

Joao Wendrich Teixeira: Investigation, Writing – review & editing.

Matty Berg: Conceptualization, Methodology, Funding acquisition, Writing – review & editing, Supervision.

James Weedon: Conceptualization, Methodology, Validation, Writing – review & editing, Supervision.

Competing interests

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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