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Key Points:

- Rewetting of a nutrient-poor boreal peatland increased the lateral export of DOC, DIC, and especially CH_4
- The expansion of open-water areas and altered hydrological flow paths were identified as key drivers for the enhanced carbon export
- Radiocarbon analysis showed a dominance of contemporary DOC being exported, with an indication of even younger DOC after rewetting

Supporting Information:

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

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Changes in Aquatic Carbon Following Rewetting of a Nutrient-Poor Northern Peatland

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Abstract Rewetting drained peatlands by raising the groundwater table is currently suggested, and widely implemented, as an efficient measure to reduce peat soil degradation and decrease CO_2 emissions. However, limited information exists regarding effects of peatland rewetting on lateral carbon export (LCE) via the aquatic pathway. Any changes in LCE are critical to consider, as they affect the overall peatland C balance, and may offset any climatic benefits from rewetting. Additionally, altered LCE could have consequences for downstream water quality and biota. Here, we monitored aquatic C content (DOC, DIC and CH_4) in runoff and pore water, as well as radiocarbon content of DOC in runoff from a drained, nutrient-poor boreal peatland that was rewetted during autumn 2020. By comparing pre- (2019–2020) and post- (2021–2022) rewetting periods, we detected changes in the aquatic C export. The results showed that the rewetting effect was site-, season- and C form-specific. Overall, one catchment showed elevated (DOC, DIC) or highly elevated (CH_4) concentrations and exports post-rewetting, whereas the other site showed only elevated DOC. Changes in runoff C concentrations after rewetting were likely driven by site-specific factors such as expansion of open-water areas, altered hydrological flow paths and proportion of filled in ditches of total ditch length. Finally, radiocarbon measurements indicated enhanced export of contemporary DOC via runoff following rewetting. These initial (short-term) findings highlight the need for site-specific before-after assessments to better evaluate the C sequestration capacity of peatlands while undergoing rewetting operations.

Plain Language Summary Rewetting of previously drained peatlands is currently widely conducted to reduce atmospheric carbon dioxide emissions. However, little is known about how rewetting affects the transport of carbon through water, known as lateral carbon export (LCE). Changes in LCE are critical to consider because they can affect the carbon balance of peatlands, while also affecting downstream water quality and associated ecosystems. In this study, we monitored aquatic carbon concentrations and age in the outlets of a nutrient-poor boreal peatland that was rewetted in autumn 2020. By comparing data from before (2019–2020) and after (2021–2022) rewetting, we found that rewetting increased concentrations and water exports of all major forms of carbon and especially methane, a powerful greenhouse gas. Changes in the concentrations of carbon at the outlets were likely affected by factors like the expansion of open-water areas and altered hydrological configuration upon rewetting. Finally, we found that younger dissolved organic carbon was exported after rewetting. These short-term results, observed after 2 years, highlight the need for site-specific monitoring before and after rewetting to better understand the carbon sequestration potential of peatlands and to avoid overestimating the climate benefits of these operations.

1. Introduction

Peatlands cover only 3% of the Earth's surface (Xu et al., 2018) but are estimated to store 20%–30% of the global soil carbon (C) (Gorham, 1991; Scharlemann et al., 2014; Yu et al., 2010), thus representing an important role in regulating the atmospheric C content and hence the climate. For forestry intensive regions like the Nordic-Baltic countries, vast areas of peatland soils (almost 10 million ha) have historically been drained by man-made ditching to increase wood biomass production (Norstedt et al., 2021; Päivänen & Hånell, 2012; Strack, 2008). Sweden has one of the most drained forested landscapes globally (Peatland and Climate Change, 2023), with drained peatlands accounting for 14% of its productive forests (Hånell, 2009). Large peatland areas have also remained unproductive despite drainage, mainly due to nutrient limitation (Hånell, 1988; Sikström & Hökkä, 2016). These extensive drainage efforts have led to degradation of peat soils through mineralization of the organic C, resulting in enhanced carbon dioxide (CO_2) emissions. To combat the increasing atmospheric CO_2 levels, efforts are now being directed toward rewetting drained peat soils as a nature-based solution to mitigate CO_2 emissions.

Rewetting previously drained peatlands to restore their natural character is typically done by blocking and filling the drainage ditch network with the aim of raising the groundwater table. The rationale from a climate mitigation perspective, is that a higher groundwater table will reduce the aerobic degradation of peat leading to decreased atmospheric CO_2 emissions. However, as more anaerobic conditions are created, the formation and emission of methane (CH_4) are expected to increase (up to 25–30 times higher, Koskinen et al., 2016), partially counteracting the climate benefits of reduced CO_2 emissions. Most empirical studies compare rewetted peatlands with drained or pristine references, assuming that differences in the C balance could be attributed to the rewetting. Recent evidence shows that both net CO_2 and CH_4 emissions could significantly increase during the first growing season post-rewetting (Laudon et al., 2023). However, these kinds of empirical studies are scarce, underscoring the need for before-and-after measurements to accurately assess rewetting impacts on the C balance.

Previous efforts to assess changes in C balance following rewetting only consider the vertical atmospheric exchange, neglecting any alterations in the lateral export of C (LCE) via runoff. Ditches in peatlands intrinsically connect the drainage network with surrounding soils and transport significant inputs of terrestrial-derived C downstream. These inflows are largely controlled by variations in hydrological inputs (Billett et al., 2006; Wallin et al., 2015), which can be altered by rewetting, along with the C source areas. The LCE typically reduces the net atmospheric C uptake in pristine boreal or temperate peatlands by 20%–50% (Dinsmore et al., 2013; Leach et al., 2016). Consequently, accounting for LCE is essential for accurate Net Ecosystem Carbon Balance (NECB) estimates and for evaluating the climate benefits of rewetting efforts (Bansal et al., 2023; Chapin et al., 2006). The LCE in pristine boreal and temperate peatlands is typically in the range of 10–30 g C m^{-2} yr^{-1} and often dominated (60%–80%) by dissolved organic carbon (DOC) in C mass (Dinsmore et al., 2013; Evans et al., 2016; Leach et al., 2016). The few existing studies on how C in boreal peatland runoff is affected by rewetting solely focus on DOC and report somewhat divergent results partly related to nutrient status (Kaila et al., 2016; Koskinen et al., 2017). The lack of data on other C forms in peatland runoff (i.e., dissolved inorganic carbon, DIC and CH_4) following rewetting prevents complete assessment of rewetting effects on the NECB. Although DIC and CH_4 typically contribute a smaller fraction of the LCE in pristine boreal peatlands, they are still a significant flux term (in mass C) that needs to be accounted for (Leach et al., 2016). Also, the higher GWP (i.e., Global Warming Potential) of CH_4 (27 times CO_2 over a 100 years horizon, according to IPCC, 2023) means that CH_4 could be a major contributor to the LCE ($\leq$50%) if expressed as CO_2 equivalents (CO_2-eq). Presently, data on how rewetting affects the LCE from boreal peatlands, particularly the export of CH_4, is lacking. With CH_4 production expected to rise in rewetted soils, there is a clear risk of overestimating the climate benefits of rewetting if lateral CH_4 export via runoff is not considered.

In addition to the mass of the LCE, any change in its C age composition is of concern when interpreting the effects of rewetting including a time dimension. Soil organic C stores in boreal peatlands have accumulated since the last glaciation (Frolking & Roulet, 2007). Efforts to radiocarbon (^{14}C) date the C from pristine boreal peatlands have found runoff C of mostly contemporary origin despite draining peat aged up to ~8,000 years old (Campeau et al., 2017). As rewetting will result in a higher groundwater table and changed groundwater flow paths, altered terrestrial C sources are to be expected as well. To what extent this will influence the age composition of the LCE is, to the extent of our knowledge, a largely unexplored, but critical factor when evaluating the rewetting effects on the peatland C balance and the stability of the soil organic C store.

Another crucial, yet scarcely explored aspect of rewetting boreal peatlands is its influence on downstream water quality. One of the most pressing water quality issues across the northern hemisphere is the brownification of many surface waters mainly caused by increased DOC concentrations (Eklöf et al., 2021; Monteith et al., 2007). Elevated DOC levels and browner waters have chemical, physical, as well as ecological consequences for downstream aquatic systems. Also, it poses a critical problem for drinking water production, which heavily relies on surface water resources (Kritzberg et al., 2020). The increasing DOC levels in many boreal surface waters in combination with the divergent results of rewetting and its influence on DOC highlight the urgent need for improved understanding of the consequences of such rewetting efforts.

To overcome these knowledge gaps, we performed a novel and detailed study on how rewetting affects runoff C chemistry of a nutrient-poor peatland in northern Sweden. We used a unique full-scale rewetting experiment designed according to a BACI approach (Before-After Control Impact), and which was sampled during a 4-year

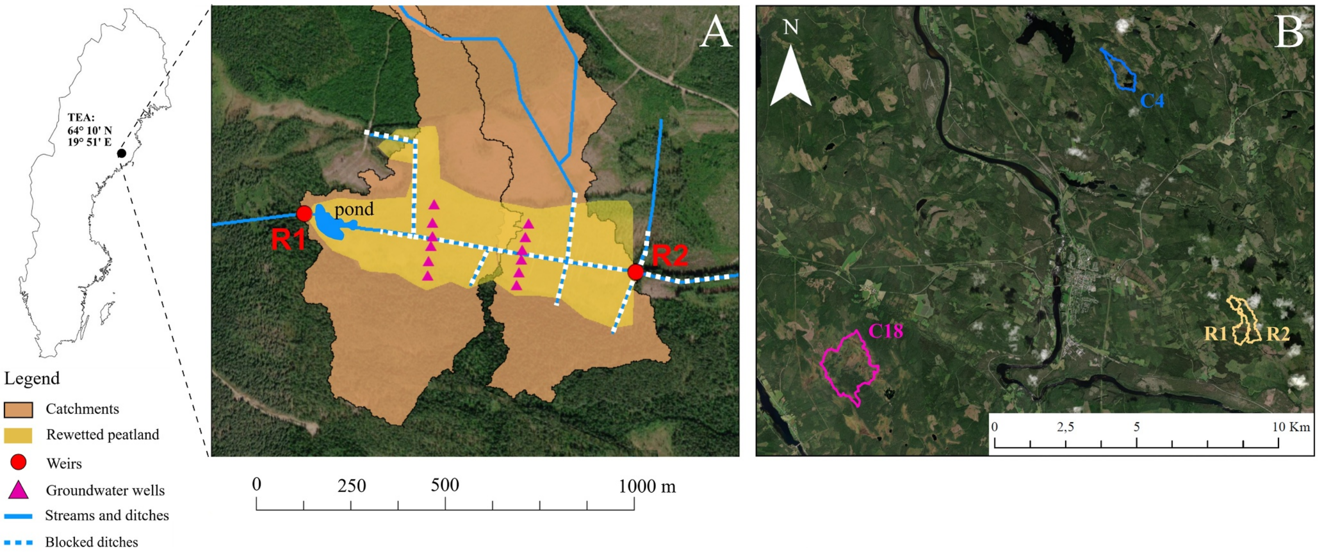

Figure 1. Panel (a) Map of the rewetted peatland at the Trollberget Experimental Area (TEA); R1 and R2 (in red) show the catchment outlets and where runoff sampling was performed. The figure shows the area after rewetting including the extended pond area upstream R1. Note that the locations of the wells are not to scale (4, 10, 20 m from ditch), but shown as displayed for clarity. Panel (b) Overview of all four study catchments, illustrating their spatial scale and relative positioning within the study area.

period (2019–2022). The aims of the study were to (a) evaluate the effects of rewetting on aquatic C (DOC, DIC and CH_4) concentrations in runoff and its hydrological control, (b) explore and quantify any changes in the LCE following rewetting, and finally, (c) trace changes in the age composition of the DOC being exported via runoff using radiocarbon analysis (^{14}C-DOC). Due to an elevated water table following rewetting and increased connectivity with organic-rich soils, we hypothesize that rewetting would enhance runoff C concentrations as well as exports of all C forms. We further hypothesize that altered intra-annual runoff patterns in response to the rewetting (Karimi et al., 2024) would result in altered hydrological controls on C mobilization. Finally, we expect that rewetting will lead to an enhanced mobilization of relatively younger DOC via runoff as more shallow soils will be hydrologically connected.

2. Study Area

The study was conducted at Stormyran, a historically drained nutrient-poor peatland located within the Trollberget Experimental Area (TEA), ca. 50 km from the city of Umeå, northern Sweden (Figure 1) (Laudon et al., 2023). TEA was established in 2018, as a part of the Krycklan Catchment Study (KCS, Laudon et al., 2021), with the aim to investigate environmental impacts of different forest management practices, including rewetting. The area of TEA is representative of the boreal biome with a mean annual air temperature of 2.1°C (30-year mean, 1986–2015), highest mean monthly temperature occurring in July (14.6°C), and the lowest in January (−8.6°C). Mean annual precipitation is 614 mm, with approximately 35%–50% falling as snow, and 311 mm contributing to runoff (Laudon et al., 2023). Typically, the area remains snow-covered for about 167 days per year, usually from late October to early May (Laudon et al., 2021).

Stormyran is located ca. 225–230 masl. It is, like many peatlands in the area, classified as an oligotrophic minerogenic mire, primarily vegetated by *Sphagnum spp.* mosses. The peatland, which covers ca. 25% of the overall catchment area, also features sparse vegetation coverage of sedges and dwarf shrubs, with occasional slow-growing Scots pine (*Pinus sylvestris*) trees. The surrounding upland forested areas are dominated by Norway spruce (*Picea abies*), with an understory of ericaceous shrubs, for example, bilberry (*Vaccinium myrtillus*) and lingonberry (*Vaccinium vitis-idaea*), growing on moss-mats composed of *Hylocomium splendens* and *Pleurozium schreberi*. The peatland soil is classified as Histosol with an average peat depth of 2.4 m (0.2–6 m). The average C/N ratio of the topsoil is 42.7 (Laudon et al., 2023). The surrounding upland forest soils are dominated by Humic Podzols with some areas of Humu-Ferric Podzols. The underlying bedrock is composed predominantly of base-poor and silicate-dominated Sveco-fennian metagraywacke and contains no known

Table 1
Catchment Characteristics of Study Sites

Catchment	Location	Area (ha)	Peat (%)	Forest (%)	Till (%)	Rock outcrops (%)	Ditch length[a] (m)	Ditch density[a] (m ha^{-1})	Length of ditches managed	% of ditches managed	Type of management
R1	TEA	47	28	54	43	18	1,986	42	677	34	Rewetted
R2	TEA	60	23	55	56	22	5,189	86	824	16	Rewetted
C4	KCS	18	40	60	17	0	50	-	-	-	Non-drained
C18	KRI	273	72	28	28	0	250	-	-	-	Non-drained

[a]Refers to period before rewetting.

carbonate minerals. The presence of nutrient-poor mineral soils and bedrock creates nutrient-limited conditions, which is evident in the species composition of the plant communities inhabiting the mire. Man-made ditching of Stormyran was conducted around 1905, and then expanded into the forested uplands of the catchment in the 1930s. The ditching was likely motivated by a combination of stimulating forest productivity and avoiding paludification of the adjacent forest. A main channel was dug centrally through the peatland in a west-east direction and with additional ditches created perpendicular to the main ditch that extended to more upland mineral soils. The ditch network drains the peatland in two directions, toward the west (R1) and east (R2) (Figure 1). Despite being part of the same mire complex, sub-catchments R1 and R2 differ in some characteristics (Table 1). In addition, R1 has an open-water pond ca. 100 m upstream of the sampling location. The total length of drainage ditches in Stormyran prior to rewetting was 1.5 km corresponding to a ditch density of ca. 128 m ha^{-1} (Table 1) including both R1 and R2. Stormyran was rewetted in November 2020 (reference date: 30 November 2020, when the operation was completed). The rewetting procedure followed conventional authority-based methods and was done by blocking the central ditch about every 50 m with log dams constructed from trees felled on-site. Most trees on the peatland were cut and removed prior to rewetting to reduce evapotranspiration losses and further restore the hydrology. The dams were further covered with geotextile to prevent mobilization of particles. The dams and the spaces between the dams were filled with on-site derived peat material. There was no active revegetation (e.g., sowing seeds or transplanting *Sphagnum*).

To assess the impact of rewetting on lateral C export, data from Stormyran were compared with two nearby, non-drained and long-term monitored peatland catchments, Kallkälsmyren (C4) included in KCS (Laudon et al., 2021) and Degerö-Stormyr (C18) included in Kulbäckslidens Research Infrastructure (KRI) (Noumonvi et al., 2023) (see details in Table 1). These two non-drained peatland catchments are comparable to the current study site and serve as controls in the analysis. The runoff outlet from the C4 sub-catchment, located ca. 10 km from Stormyran, originates from an area influenced by a nutrient-poor, minerogenic mire, covering approximately 40% of the catchment land, while the remaining portion is covered by forests. The bedrock, similarly to TEA, is predominantly composed of sedimentary veined gneiss, and overlain by glacial till of varying thickness. The peatland vegetation is primarily covered by *Sphagnum spp.* A detailed description of the site is given by Yurova et al. (2008). The second control catchment, C18, located ca. 15 km from Stormyran, is a nutrient-poor, minerogenic mire that is used as a representative of many high-latitude pristine mires. The peatland covers approximately 72% of the catchment area and is underlain by a relatively impermeable layer of mineral glacial till and gneissic bedrock. A detailed description is given by Noumonvi et al. (2023).

3. Methods

Measurements of hydrology and water chemistry in the two outlets (R1 and R2) of Stormyran started in December 2018 (Laudon et al., 2023). The sampling methodology and analytical procedures followed the already established protocols within KCS and KRI (Laudon et al., 2021; Noumonvi et al., 2023). The setup of the study followed a BACI approach with two years (2019 and 2020) of measurements during before-rewetting, and two years (2021 and 2022) during after-rewetting conditions. In this study design, years were defined utilizing the date of when the rewetting efforts were finalized, that is, 30 November 2020, as reference. Based on this pattern, each year spanned from November 30 of 1 year to November 30 of the following year.

3.1. Water Sampling and Analysis

Sampling for ditch water concentrations of DOC, DIC and CH_4 were conducted in the two outlets of Stormyran (R1 and R2) and the control catchments (C4 and C18), upstream of V-notch weirs or flumes (C18). Samples were collected every second week during summer and fall and more intensively during spring flood. Sampling during the winter (early November-early April) occurred monthly. Water samples were collected in acid-washed high-density polyethylene bottles and kept dark and cold during transport. Samples were filtered (0.45 μm mixed cellulose ester (MCE) syringe filters, Millipore®) within 24 hr and kept refrigerated at 4°C until analysis (<7 days after filtering). DOC analysis consisted of acidification of the sample for removing inorganic C, followed by combustion using a Shimadzu TOC-VCPH (Laudon et al., 2011). The particulate fraction of organic C (POC) typically exhibit a low contribution (on average <0.6%) to the total organic C (TOC) in runoff for these kinds of systems (Laudon et al., 2011). Thus, DOC is considered equal to TOC. Samples for DIC and CH_4 were collected by injecting 5 mL of ditch water with a syringe into a sealed 22.5 mL glass vial. The vials were evacuated prior to sampling, flooded with N_2 at atmospheric pressure and prefilled with 0.1 ml 85% H_3PO_4 to shift the carbonate equilibrium toward CO_2. Headspace CO_2 and CH_4 concentrations were analyzed on a gas chromatograph equipped with a methanizer and flame ionization detector (GC-FID). In situ ditch concentrations of DIC and CH_4 were calculated from headspace concentrations considering water and headspace volumes and temperature-dependent equations. In our measurements, DIC includes CO_2, HCO_3^- and CO_3^{2-}, however, in low pH waters like these (average pH = 4.7), DIC mainly consists of CO_2 (Stumm & Morgan, 1996). For further details concerning DIC and CH_4 sampling and analysis see Wallin et al. (2010, 2014) and Åberg and Wallin (2014). Sampling for DOC, DIC and CH_4 in shallow pore water (<1 m depth) was conducted in two 40 m long transects (one in each catchment) distributed perpendicular to the main ditch channel (Figure 1). Each transect consisted of six groundwater tubes installed 4, 10 and 20 m from each side of the ditch. The tubes were emptied prior to sampling to prevent stagnant water contamination and to ensure collection of representative pore water. Pore water chemistry sampling was conducted on four occasions during the study period, one before (09-2020) and three after the rewetting (06-2021, 10-2021 and 10-2022) for the same variables as sampled in the ditches. Samples for DIC and CH_4 were collected two out of three times after rewetting.

3.2. Radiocarbon Sampling

Sampling for ^{14}C content in runoff DOC was conducted at the two rewetted sites (R1 and R2) and at the two control sites (C4 and C18) on 10 occasions, three times before (04, 08, 10-2020) and seven times after rewetting (04, 08, 09, 10-2021 and 05, 08, 10-2022). Grab samples were collected in 500 mL polypropylene sample bottles that had been acid-washed and pre-rinsed three times with sample water prior to sampling. Samples were filtered in a laboratory on the same day through pre-baked glass-fiber filters (GF/F, 0.7 μm). Filtered samples were kept dark and cold and shipped to the National Environmental Isotope Facility Radiocarbon Laboratory (East Kilbride, UK) where they were freeze-dried to solids. The samples were then acid-fumigated and combusted to CO_2 using the sealed quartz tube method (Ascough et al., 2024). The CO_2 was cryogenically purified and split into aliquots with one part analyzed for $\delta^{13}C$ using isotope ratio mass spectrometry (Delta V, Thermo-Fisher) and used to normalize the ^{14}C results to a $\delta^{13}C = -25‰$ to correct for isotopic fractionation. A second aliquot was converted to graphite using Fe: Zn reduction and measured for ^{14}C content using accelerator mass spectrometry (AMS) at the Scottish Universities Environmental Research Centre AMS Facility. Radiocarbon results are reported as % modern C (where %modern C = fraction modern x 100) and conventional radiocarbon years (in years BP, where 0 BP = 1950 CE).

3.3. Hydrological Measurements

Water discharge was measured at the outlet of each catchment (Karimi et al., 2024). Measurements were performed in V-notch weirs (R1, R2, C4) or a flume (C18) using established stage height-discharge relationships. Discharge gauging for rating curve definition was done using time-volume measurements (R1, R2, C4) or salt dilution (C18) covering most of the observed discharge ranges (Karimi et al., 2024). Stage height was continuously recorded at each site using capacitance (Tru-track WT-HR 500 at R1 and R2) or pressure (MJK at C4 and C18) sensors. Discharge data for R1 and R2 sites during the period 1 December 2018, to 27 April 2019, were missing. The missing discharge values were estimated based on the relationship observed between discharge data from each of the R sites with the average of the controls during corresponding time spans in the following years.

The average groundwater table at the Stormyran site was 216 mm below ground before the rewetting (in 2020) and increased by 131–85 mm below ground 2 years after the rewetting (in 2022). In contrast, the average groundwater level at the control site decreased by 2.5 mm during the same period. The rewetting effects on the groundwater table also varied between the rewetted catchment, but with relatively small differences between R1 (81 mm difference in average levels before and after rewetting) and R2 (59 mm) (Karimi et al., 2024, unpublished data). For further information on the hydrological effects from the rewetting, see Karimi et al. (2024).

3.4. Calculations and Statistical Analyses

Statistical differences in C concentrations and ^{14}C-DOC content in runoff were tested using a Generalized Linear Mixed Model (GLMM). Absolute runoff C concentrations and ^{14}C-DOC content were Log-transformed to achieve a normal distribution. Differences between pre- and post-rewetting conditions were evaluated by comparing the relative differences in measurements at rewetted and control sites, specifically measured 2 years before and 2 years after the rewetting. The relative difference (Δ) of mean log C concentration for the two rewetted sites, R1 and R2, versus the mean values of the two reference sites, C4 and C18, was calculated for each sampling occasion. This approach assessed the magnitude of concentration variation at the rewetted sites compared to their corresponding controls. In the GLMM, before or after rewetting were used as a fixed factor, and a repeated structure (date) accounted for the repeated sampling at each site. When dividing analyses in seasons, these were defined as follows: Spring (April–June), Summer (July–September), Autumn (October–December), and Winter (January–March). No similar GLMM tests were done on pore water C concentrations due to limited and also unevenly distributed data for the periods before and after rewetting. Differences in pore water C concentrations between the R1 and R2 catchments were tested using the non-parametric Wilcoxon test.

The response in C concentrations to variable discharge was analyzed by constructing concentration-discharge (C-Q) relationships [Log [C] (mg L^{-1}) versus Log specific discharge (mm d^{-1})]. Such C-Q plots were created for each study site to assess whether the hydrological control was changing in response to the rewetting. The slope values obtained from the C-Q regressions for the period before and after rewetting were interpreted following Meybeck and Moatar (2012). Sites were considered "source limited" when the slope <-0.2, "chemostatic" when the slope ranged between -0.2 and 0.2, and "transport limited" when the slope was ≥ 0.2. Any significant changes in the slopes of the C-Q regressions before and after rewetting were evaluated using an ANCOVA approach. The ANCOVA model was constructed with Log-Q as the covariate, treatment time as the categorical factor, and the interaction term between Log-Q and treatment time to test for differences in C-Q slopes between the pre- and post-rewetting periods.

Aquatic C exports (expressed in g C m^{-2} d^{-1} or g CO_2-eq m^{-2} d^{-1} (considering a GWP of CH_4 of 27 times that of CO_2, according to IPCC, 2023)) were estimated for each C form by multiplying the mean daily discharge by daily C concentration. Daily time series of aquatic C concentrations were derived by a linear interpolation between sampled data. Daily exports were further summed per year and divided by catchment area.

All data and statistical analyses were performed using JMP Pro 17 (SAS Institute Inc., Cary, NC, USA) with a significance level of $p < 0.05$.

4. Results

4.1. Runoff C Concentrations Following Rewetting

There were clear differences in runoff C concentrations between the two rewetted sites, R1 and R2 (Figure 2). Mean DOC concentration was higher in R2 than in R1 before (33.4 and 25.2 mg L^{-1}, $p < 0.0001$) and after (34.4 and 29.2 mg L^{-1}, $p = 0.001$) rewetting. In contrast, mean DIC concentration was higher in R1 than in R2 before (8.3 and 5.4 mg L^{-1}, $p = 0.0004$) and after (9.4 and 4.4 mg L^{-1}, $p < 0.0001$) rewetting. Similarly, mean CH_4 concentration was higher ($p < 0.0001$) in R1 than in R2 before (284.9 and 58.0 μg C L^{-1}) and after (996.7 and 35 μg C L^{-1}) rewetting.

At R1, ΔDOC concentrations increased after rewetting compared to before (Figure 3a). Mean ΔDOC concentration at R1 was 8.4 mg L^{-1} lower than the control sites before rewetting, whereas it was similar to the controls (0.5 mg L^{-1} lower) after (Figure 3a). At R2, ΔDOC also increased after rewetting (Figure 3b). Mean ΔDOC concentration was 0.8 mg L^{-1} higher compared to control sites before rewetting, while after rewetting, it was 5.2 mg L^{-1} higher (Figure 3b). ΔDIC concentrations increased at R1 after rewetting (Figure 3c), but not at R2

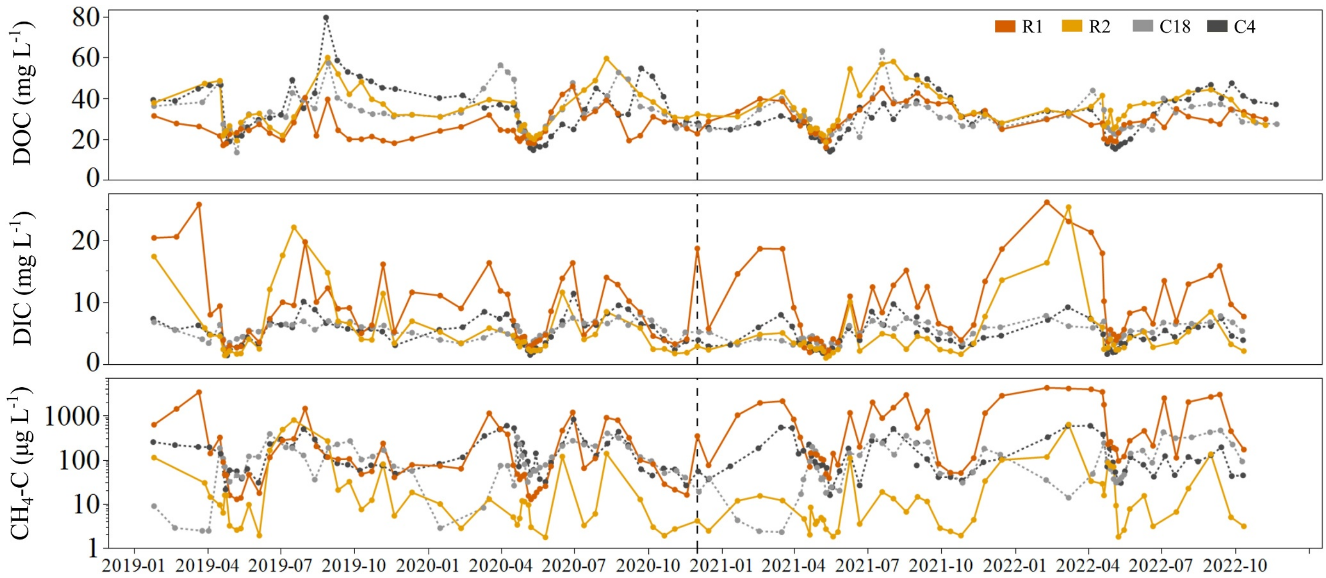

Figure 2. Time series of DOC, DIC and CH_4 concentrations for the two outlets of the peatland at TEA (R1 and R2) and their controls C4 and C18. The vertical dotted line indicates the reference date (30 November 2020) when rewetting was conducted. Note the Log scale on the *Y*-axis for CH_4-C concentrations.

(Figure 3d). In relation to the controls, mean ΔDIC were 2.6 and 4.4 mg L^{-1} higher at R1 before and after rewetting, while 0.1 mg L^{-1} lower and 0.4 mg L^{-1} higher before and after rewetting, respectively, at R2 (Figures 3c and 3d). ΔCH_4 concentration in the outlet of R1 massively increased (989%, $p < 0.0001$, Figure 3e) after rewetting, but no change was observed at R2 (Figure 3f). In relation to the controls, mean ΔCH_4 at R1 was 74.7 and 813.4 µg C L^{-1} higher before and after rewetting, respectively (Figure 3e), while it was 89.1 and 70.9 µg C L^{-1} lower at R2 (Figure 3f).

4.2. Seasonal and Flow-Dependent Impacts on Runoff C Concentrations Upon Rewetting

On a seasonal scale, ΔDOC concentrations increased post-rewetting compared to pre-rewetting at R1 during summer, autumn and winter, but showed no change in spring (Figure 4a). At R2, ΔDOC concentrations increased post-rewetting during spring and winter, while remaining unchanged during the other seasons (Figure 4b). An increase in ΔDIC concentrations was observed at R1 post-rewetting, but only during autumn (Figure 4c). In contrast, ΔDIC decreased post-rewetting at R2 during summer (Figure 4d). ΔCH_4 concentrations increased post-rewetting at R1 during all seasons (Figure 4e), whereas no change in ΔCH_4 was observed at R2 post-rewetting, regardless of season (Figure 4f).

Before rewetting, significant negative log-transformed concentration-discharge (C-Q) relationships were observed at all study sites for all C forms, with the exception of CH_4 at site C18 (Figure 5, Table S1 in Supporting Information S1). The explanatory power varied across sites and C form (R^2, 0.14–0.82). The slopes of the C-Q relationships indicated a source-limited behavior (slope <-0.2) for both DIC and CH_4 at sites C4, R1, and R2, while relationships were chemostatic for DOC at all sites and for DIC at site C18 (Table S1 in Supporting Information S1).

After rewetting, there was no significant change in the hydrological (discharge) control on C concentrations. Although some of the negative slopes for the log-transformed C-Q relationships showed a tendency to steepen post-rewetting (e.g., for DIC-Q: from −0.32 to −0.40 at R1 and from −0.24 to −0.33 at R2; for CH_4-Q: from −0.60 to −0.74 at R1 and from −0.36 to −0.50 at R2; Table S1 in Supporting Information S1), these changes in slopes were not statistically significant.

4.3. Pore Water C

In contrast to what was observed in runoff, overall mean pore water DOC (50.0 and 49.1 mg L^{-1} at R1 and R2, respectively) and CH_4 (1,224.6 and 1,469.0 µg L^{-1}) concentrations did not differ between the two catchments R1

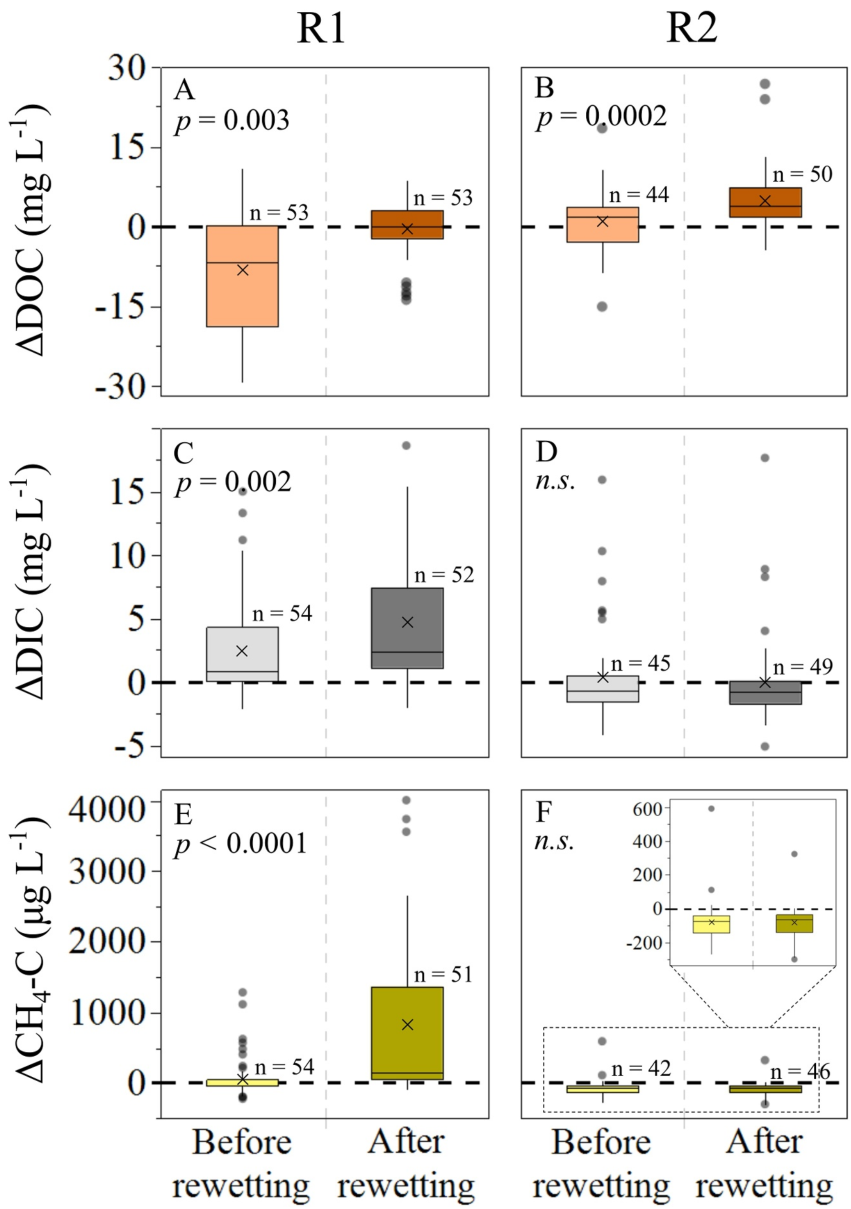

Figure 3. Differences between the treatment and control sites before and after rewetting for concentrations of DOC, DIC and CH_4 at R1 and R2. The difference (Δ) was calculated as treatment minus control, thus positive values indicate higher C concentration at treatment site compared to the control site, while negative values indicate the opposite. Mean Δ values are shown with a "×." Statistically significant differences from the mixed models are reported with corresponding *p*-values or "*n.s.*" if not significant. Note that actual (non-Log-transformed) data are shown on *y*-axis.

and R2 ($p = 0.54$ and $p = 0.22$, Figures 6a–6b, 6e–6f). Mean DIC concentration was higher in R2 (14.4 mg L^{-1}) than in R1 (10.0 mg L^{-1}) ($p = 0.0005$, Figures 6c and 6d), which was opposite to what was observed in runoff (see Section 4.1). Further detailed information of pore water C concentrations within the transects are shown in Figure S1 in Supporting Information S1.

4.4. Changes in Lateral C Export

In all study catchments and for the whole study period, the LCE was dominated by DOC (mean: 86%), followed by DIC (14%) and CH_4 (0.4%). The relative proportion of different LCE forms (DOC, DIC and CH_4) before and after rewetting did not change at the controls, while showed differences at the rewetted sites. At R1, the DOC:DIC:CH_4 relative composition changed upon rewetting from 83:17:0.3 (%) to 77:21:2 (%), while at R2 from

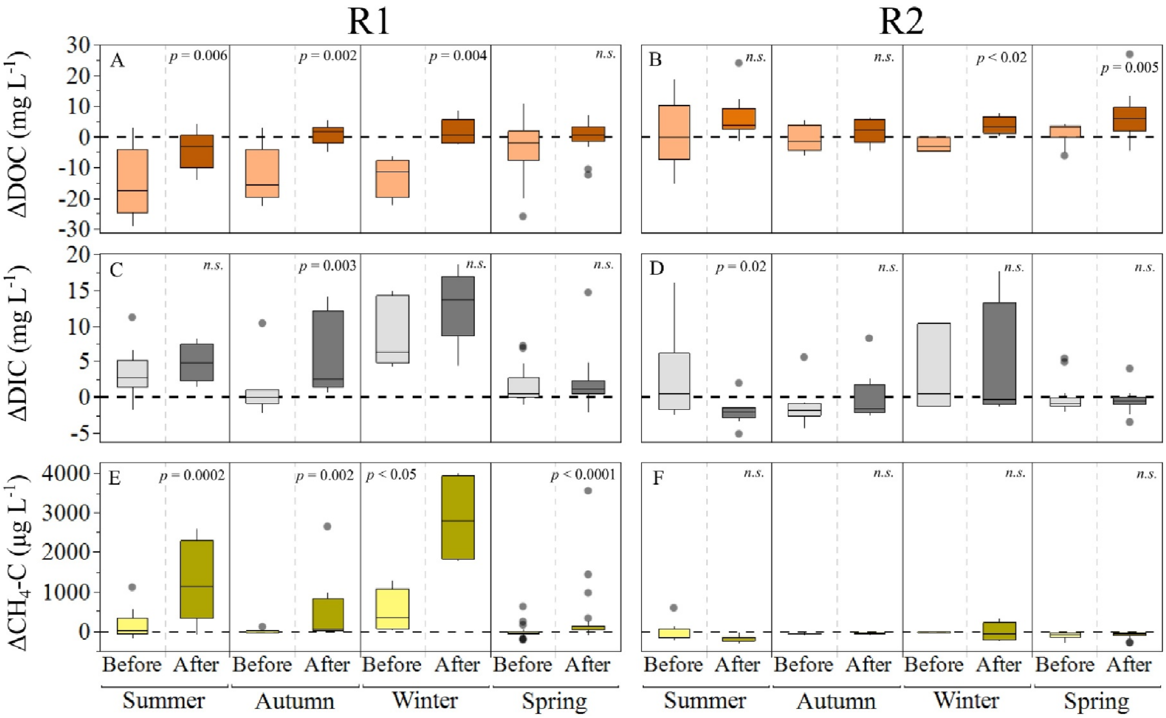

Figure 4. Seasonal differences between the treatment and control sites before and after rewetting for concentrations of DOC, DIC and CH_4 at R1 and R2. The difference (Δ) was calculated as treatment minus control, thus positive values indicate higher C concentration at treatment site compared to the control site, while negative values indicate the opposite. Statistically significant differences from the mixed models are reported with corresponding *p*-values or "*n.s.*" if not significant. Note that actual (non-Log-transformed) data are shown on *y*-axis.

87:13:0.1 (%) to 91:9:0.1 (%). The DOC exports in the period after rewetting were higher than before rewetting at both controls (~17%) and rewetted sites (64% and 13% at R1 and R2, respectively) (Figures 7a and 7b). At the rewetted sites, average DOC export increased from 6.9 to 11.4 g C m^{-2} y^{-1} (R1) and from 5.8 to 6.6 g C m^{-2} y^{-1} (R2) before and after rewetting. The DIC exports were about 20% higher at both control sites (C4 and C18) in the period after rewetting compared to the period before, while DIC exports increased markedly by 121% at R1 but decreased by 27% at R2 (Figures 7a and 7b). Average DIC exports for the rewetted sites increased from 1.4 to 3.0 g C m^{-2} y^{-1} at R1, yet decreased from 0.8 to 0.6 g C m^{-2} y^{-1} at R2. The average CH_4 export at both control sites (C4 and C18) was about 35% higher for the period after rewetting compared to before (Figures 7a and 7b). The average CH_4 export at R1 increased by ~770% after rewetting compared to before. In contrast, the average CH_4 export at R2 decreased by ~60% after rewetting compared to before. At the rewetted sites, before and after rewetting, average CH_4 exports increased from 0.02 to 0.3 g C m^{-2} y^{-1} at R1, but decreased from 0.01 to 0.004 g C m^{-2} y^{-1} at R2.

In terms of CO_2-eq (Figure 7b), the impact of rewetting was particularly notable at R1, but not at R2. The large increase in LCE as CO_2-eq at R1 was primarily driven by CH_4, which increased from 2.6 to 26.3 g CO_2-eq m^{-2} y^{-1} after rewetting. Further details on inter-annual variations in LCE for each C species can be found in Table S2 in Supporting Information S1.

4.5. Changes in ^{14}C-DOC Content Following Rewetting

The radiocarbon analysis showed that the laterally exported DOC was predominantly of post-bomb origin (i.e., ^{14}C-DOC > 100%modern), containing C fixed from the atmosphere after 1955, regardless of whether it was from pre- or post-rewetting (Figure 8). Before rewetting, mean ^{14}C-DOC content for the controls (106.1%modern (range: 104.0–107.9)) was higher ($p = 0.03$) than for the treatment sites (103.0%modern (99.5–105.9)). In the period after rewetting, C4 showed higher ^{14}C-DOC (mean: 107.5%modern) compared to before (mean: 105.3% modern) (Figure 8a). An indication of mobilization of aged DOC was detected in one DOC sample collected at R1 after rewetting (89.5%modern, corresponding to 894 ± 35 years BP), but overall, the rewetting did not cause any

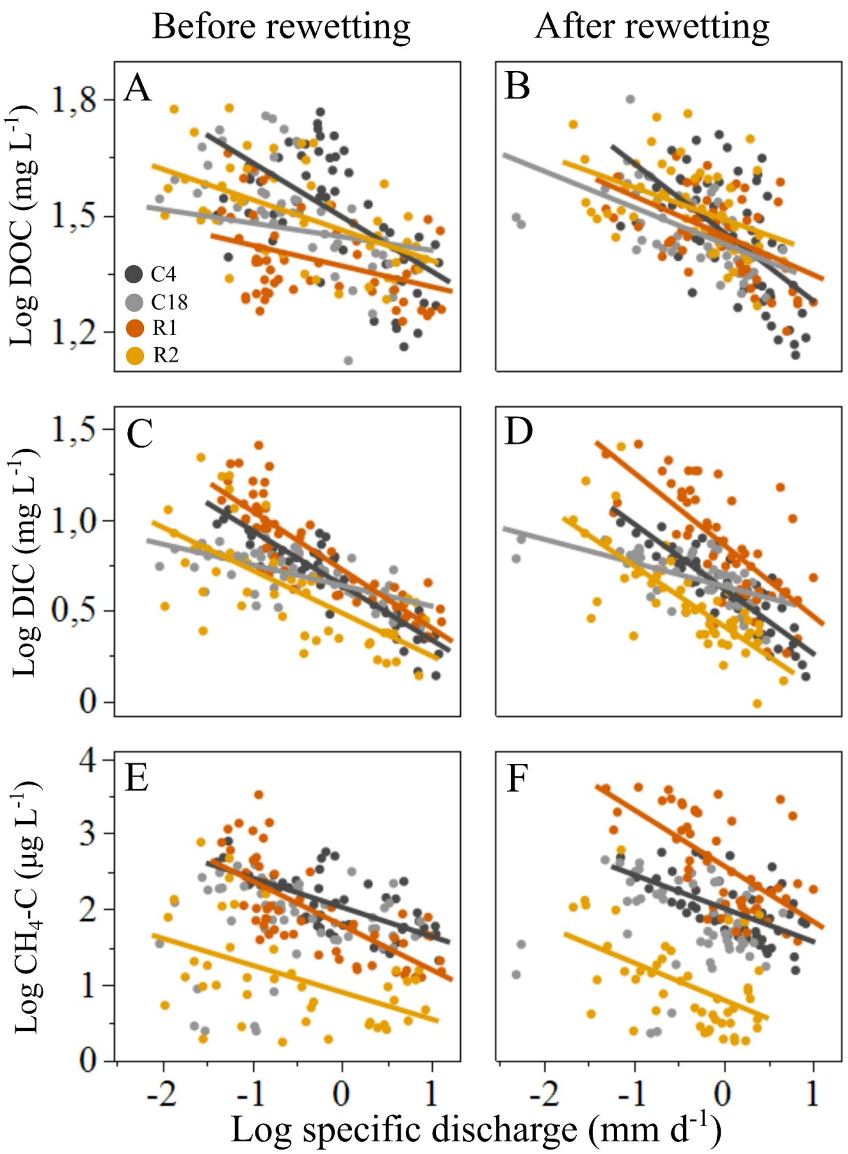

Figure 5. Log concentrations of DOC, DIC and CH_4 at the R sites and controls as a function of Log specific discharge (mm d^{-1}) for the periods before and after the rewetting. Regression lines are shown only when significant. The details of the regressions are reported in Table S1 in Supporting Information S1.

significant change in the ^{14}C-DOC content at R1 (Figure 8c). The ^{14}C-DOC content at R2 was significantly enriched post-rewetting (mean: 106.3%modern) compared to the period before (102.1%modern) (Figure 8d). Two samples collected at R2 showed an indication of relatively older (pre-bomb) DOC (^{14}C-DOC content, 99.5 and 99.8%modern), but in contrast to the sample collected at R1 these were collected in the period before rewetting. Detailed information on radiocarbon analysis results is summarized in Table S3 in Supporting Information S1.

5. Discussion

5.1. Rewetting Effects on Concentrations, Pore Water-Runoff Linkages, and Exports of C

Supporting the main hypothesis, the results from R1 showed that rewetting of a nutrient-poor boreal peatland significantly increased the total LCE via runoff. Hence, ignoring this enhanced C flux in the overall assessment of the NECB could lead to an overestimation of the climate benefit of rewetting. In contrast, we found no change in

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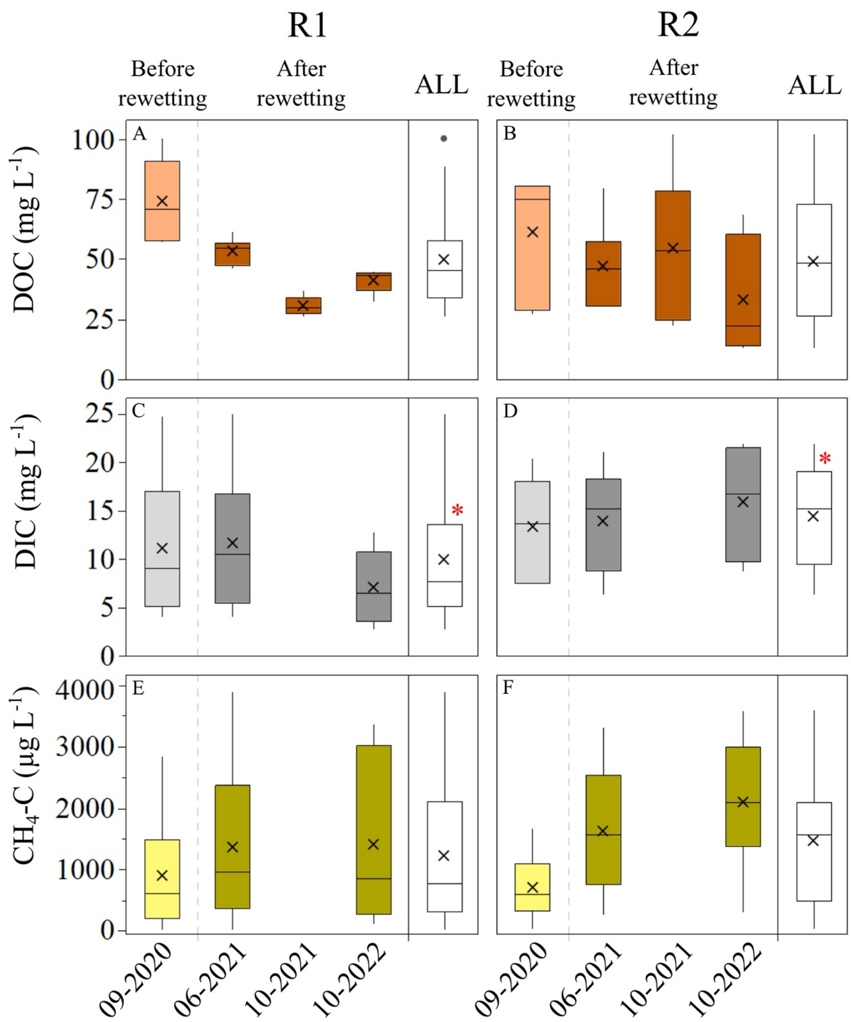

Figure 6. Box plots of pore water concentrations of DOC, DIC and CH_4, along the 40 m transects in the R1 and R2 catchments, shown for each sampling occasion before and after rewetting. Each box plot represents six measurements, one from each well along the transect. An additional box plot is included to show all measurements combined, without accounting for the effect of rewetting. Mean values are marked with an "×." Red stars "*" indicate significant differences based on the non-parametric Wilcoxon test ($p < 0.05$).

total LCE from the R2 catchment, highlighting a significant between-site variability that needs to be considered when rewetting effects are evaluated.

The rewetting effect on the annual LCE was largely controlled by changes in runoff C concentrations (Figure 3). There is limited literature to compare with on the effect of rewetting on total runoff C (i.e., including DOC, DIC and CH_4) in a boreal context. Koskinen et al. (2017) found elevated DOC levels in runoff from rewetted mesotrophic peatlands in boreal Finland and explained the increase through reduced oxygen conditions, which in turn breaks down iron-organic bonds in the peat, resulting in an enhanced DOC release. Studies from temperate systems further suggest that higher groundwater levels following rewetting could enhance the activity of extracellular phenol oxidases, which drives decomposition of organic material in low-oxygen conditions (Fenner et al., 2011). In line with those explanations, Menberu et al. (2017) found a substantial initial increase in pore water DOC in the first year after rewetting oligotrophic peatlands in boreal Finland. The authors further noted that the elevated DOC levels in pore water progressively decreased during a 5-year period before reaching background levels. The pore water C measurements in the current study were not designed to trace changes in relation to rewetting, still the limited number of sampling occasions did not indicate an increase in pore water DOC as was observed in runoff (Figures 6a and 6b). Also, while DOC concentrations in the R1 and R2 catchments were close

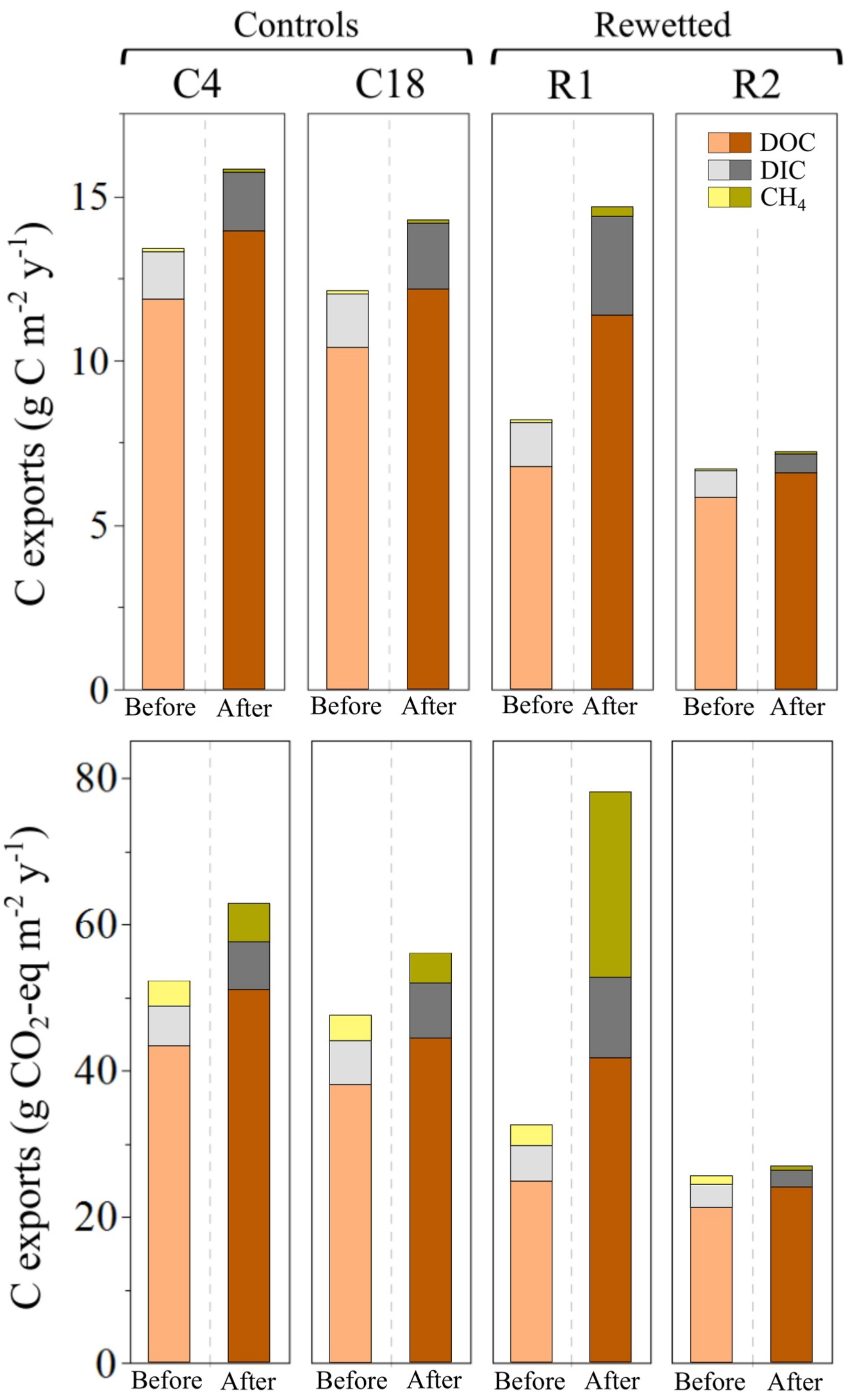

Figure 7. Yearly export of DOC, DIC and CH_4 from the control sites (C4, C18) and rewetted sites (R1, R2) before and after rewetting based on averages of the two respective years. Upper panel shows yearly exports in g C m^{-2} y^{-1} and lower panel shows the yearly exports expressed in CO_2-equivalents (g CO_2-eq m^{-2} y^{-1}).

to identical in pore water, they differed clearly in runoff, suggesting additional C sources than shallow pore water to sustain runoff. It should be noted that R1 and R2 differed in the proportion of filled in ditches of total ditch length (Table 1), with a lower percentage for R2, which could partly explain the differences in runoff C between the two catchments. We further suggest that the elevated runoff DOC concentrations and thus enhanced export following rewetting were a result of alterations in groundwater flow paths and connectivity between peat soils and ditches. The rewetting caused a shift from a system with a high and spatially distributed terrestrial-aquatic

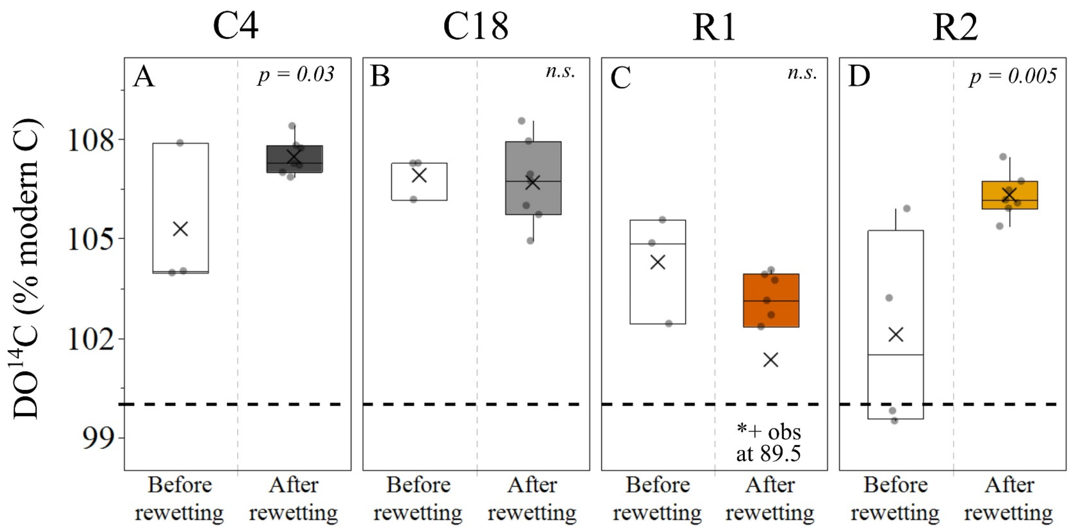

Figure 8. Distributions in ^{14}C values of DOC (%modern) in runoff of the study sites separated in the periods before and after rewetting. Note that no treatment occurred at C4 and C18 and these served as controls. Modern ^{14}C-DOC values plot above the dotted line. Statistically significant differences from the mixed models are reported with corresponding *p*-values or "*n.s.*" if not significant. Mean values are shown with a "×."

connectivity due to a dense ditch network, to one where the connectivity horizontally and vertically converges toward a newly created ditch initiation point close to the catchment outlets. Consequently, water is forced to travel longer distances through the peat and potentially also through different soil strata resulting in an altered chemical signal in runoff (Laudon & Sponseller, 2018).

For DIC, the higher overall concentrations in runoff for R1 than in R2 was not reflected in the pore water, where an opposite pattern was observed with higher concentrations in R2 (Figures 2 and 6c–6d). Also, the rewetting increased runoff DIC (both concentrations and export) at R1 but not in R2. A similar mismatch between pore water and runoff was also observed for CH_4, with no difference in average pore water CH_4 between R1 and R2 (Figures 6e and 6f), whereas runoff CH_4 concentrations were on average more than 10 times higher in R1 than in R2 (Figure 2). Similar to DIC, R1 was the only site that increased in runoff CH_4 (both concentrations and export) following rewetting. Collectively, this suggests that catchment-specific characteristics cause the different patterns observed for DIC and CH_4, and we believe the peatland pond just upstream of the outlet in R1 could be a main reason. Peatland ponds are known hotspots for GHG formation (both CO_2 and CH_4) due to their intrinsic characteristics, including shallow waters and high sediment (peat)-to-water interfaces (Holgerson & Raymond, 2016). The combination of high light penetration, warm temperatures in shallow waters, and anoxic conditions fosters microbial and photochemical degradation processes, which actively transform and decompose the substantial organic C inputs from adjacent peat soils into CO_2 or CH_4 (Arsenault et al., 2024; Dean et al., 2024; Olid et al., 2021). The pond also expanded significantly in size in response to the rewetting (roughly 300%), further increasing its contribution to DIC and CH_4 in runoff. Although it is not possible to assess by our pore water sampling design, shallower groundwater tables in response to rewetting typically enhance the extent of anoxic conditions and stimulates CH_4 production and atmospheric emission from peatland surfaces (Koskinen et al., 2016), something that has also been documented for the Trollberget site in response to the rewetting (Laudon et al., 2023).

The LCE increased substantially at R1 in response to rewetting (Figure 7). In contrast, it was largely unchanged at R2 and was approximately 50% lower than that at R1, demonstrating spatial heterogeneity across peatland systems. The average LCE for the 2 years after rewetting at R1 aligned with earlier findings from the control sites of our current study (Leach et al., 2016; Wallin et al., 2013), but was lower than typical export rates observed in pristine boreal and temperate peatlands (Billett et al., 2004; Dawson et al., 2002). As expected for this kind of boreal peatland, DOC was the most contributing C form to the LCE. R2 indicated a minor (but non-significant) increase in DOC export after rewetting, but showed overall notably lower DOC exports compared to R1, to other boreal peatlands (Rantakari et al., 2010) and to the controls (Leach et al., 2016; Wallin et al., 2013). The increase in DOC export in R1 following rewetting (+4.6 g C m^{-2} y^{-1} or 68%) mirrors the findings of Koskinen et al. (2017) for a rewetted oligotrophic mire in Finland, but was lower than the initial (i.e., first year post-rewetting) increase (about +20 g C m^{-2} y^{-1}) observed in a more fertile, rewetted mesotrophic peatland of the same Finnish study. The higher relative increase in DIC exports (120%) observed at R1 than in the controls (20%) suggests that rewetting also played a key role for enhancing DIC exports. The DIC exported via runoff in R1 after rewetting (ca. 3.0 g C m^{-2} y^{-1}) was relatively high compared to export rates found in the literature for different peatlands (Dinsmore et al., 2010; Leach et al., 2016; Rantakari et al., 2010; Rehn et al., 2023). Although representing a minor fraction of the LCE, CH_4 export increased significantly (~770%) at R1 after rewetting, but not at R2, where it decreased (~60%). The CH_4 export at R1 after rewetting was one order of magnitude higher than from R2 and the control catchments (see also Leach et al., 2016), but also relative to other studies of temperate peatlands in Scotland (Dinsmore et al., 2013; Hope et al., 2001). Given the 27 times higher GWP for CH_4 compared to CO_2 (IPCC, 2023), this increase had a substantial impact on the export accounted as CO_2-eq. The elevated CH_4 export at R1 suggests higher atmospheric emissions from downstream aquatic systems (Wallin et al., 2018), potentially offsetting terrestrial CO_2 sequestration benefits from rewetting. The increase in CH_4 export at R1 after rewetting was driven by the massive concentration increase, with post-rewetting concentrations (mean and max: 997 and 4,184 μg C L^{-1}) within the upper range found for rivers globally (Stanley et al., 2023).

5.2. Altered Seasonal Patterns and Hydrological Controls on Aquatic C Upon Rewetting

The slopes of the C-Q relationships for DOC, DIC and CH_4 were not significantly different for the periods before and after rewetting for any of the rewetted and control sites suggesting that the hydrological source control was unaffected by the rewetting. Despite showing negative slopes, the DOC-Q relationships were found to be chemostatic at all sites both in the periods before and after rewetting. This suggests a relatively stable DOC source which is weakly related to hydrological variations (Fork et al., 2020). In contrast to DOC, the source-limited C-Q patterns found for DIC and CH_4 suggest that their production and supply cannot keep up with enhanced mobilization at higher runoff. Although the C-Q slopes remained statistically unchanged, the regression offsets (primarily at R1) increased, suggesting higher base flow concentrations following rewetting (Figure 5). The enhanced baseflow DIC and CH_4 concentrations at R1, again, point toward the increased importance of the pond upstream of the outlet as the main source.

We further hypothesized that the rewetting response on runoff C concentrations would be different across seasons. It was evident that the responses were highly variable across the two sites and for the different C forms, but with few seasonally consistent patterns (Figure 4). The only season-specific rewetting related change in runoff C concentration that was observed at both R1 and R2 was an increase in ΔDOC during winter. Elevated DOC concentrations under the frozen surface of peatlands have been found and described by Ågren et al. (2012) as a freeze-out effect. In this process, DOC is excluded from the forming ice and concentrated in the liquid water below the frozen soil layer. A rewetting-induced shallower groundwater level during freezing could potentially amplify the freeze-out effect and, together with changed groundwater flow paths, result in higher runoff DOC concentrations. Runoff CH_4 concentrations increased at R1 after rewetting, regardless of season (Figure 4). The pronounced rewetting effect during summer could be connected to the temperature sensitivity of CH_4 production (Yvon-Durocher et al., 2014). More surprising was the even stronger rewetting effect on runoff CH_4 observed during the winter period. As we believe that the pond is an important CH_4 source at R1, the strongly elevated concentrations could be an effect of increased production associated with the increased pond size and enhanced anoxic conditions. Additionally, the ice cover during winter efficiently prevents atmospheric degassing, resulting in a CH_4 accumulation available for runoff (Karlsson et al., 2013).

5.3. DOC Age Following Rewetting

The radiocarbon analysis showed that mostly modern (i.e., post-bomb) DOC, within the range of ^{14}C content reported from different boreal peatlands was exported from both the rewetted and control catchments (Campeau et al., 2017, 103.5–112.2% modern; Billett et al., 2012, mean 108.4%modern). The runoff ^{14}C-DOC content at R2 was significantly higher in the period after the rewetting compared to the period before and was then more similar to the two control catchments. We attribute the change at R2 to the rewetting, suggesting an enhanced contribution of more recently fixed C. This finding supports the hypothesis that peatland rewetting causes mobilization of more contemporary DOC from shallow peat layers which become more hydrologically connected. However, a similar significant change in ^{14}C-DOC content was also visible in one of the control sites (C4) suggesting that intra-annual variability in the DOC mobilization could have partly influenced the observed change at R2. In contrast, no significant difference in ^{14}C-DOC content was observed for the periods before and after rewetting at R1. One aged sample (89.5%modern, corresponding to 894 ± 35 years BP) was collected in the period after the rewetting. Whether this single sample of aged DOC could be caused by the rewetting is hard to know but cannot be ruled out. Very few studies have focused on determining the age of the LCE after different types of management practices including rewetting of drained peatlands. Butman et al. (2015) suggested that drainage intensity and disturbed riparian zones likely promote the mobilization of aged soil C, but stressed the need for more focused research. Campeau et al. (2019) found a positive correlation between ^{14}C-DOC and the water table level in organic-rich riparian areas of boreal headwater streams. Our findings at R2 are consistent with those previous results, despite that the Campeau et al. (2019) study was focusing on a pristine forested catchment rather than a rewetted peatland. Hydrological activation of more superficial soil layers at higher groundwater tables mobilize recently degraded DOC with a more modern ^{14}C signal, modifying the character but not necessarily the amount that becomes available for runoff.

5.4. Conclusions and Future Research Needs

This study presents a complete dissolved C (DOC, DIC and CH_4) assessment of lateral carbon export (LCE) runoff concentrations as well as export rates before and after rewetting of a drained nutrient-poor boreal peatland. The BACI approach, with data collection 2 years before and 2 years after rewetting in both treated and control catchments provides empirical data which is currently scarce on the topic (Escobar et al., 2022). Although site-specific patterns were observed, a strong increase in the LCE of all C forms was observed in one of the two sites following rewetting. Notably, the lateral CH_4 export, expressed as mass of C, increased significantly following rewetting. When converted to CO_2-eq using the GWP of CH_4 (27 times that of CO_2 over 100 years), CH_4 became a major component of the total LCE in terms of its climate impact, making it a critical factor for GHG balance estimates. Moreover, from a water quality perspective, rewetting of drained peatlands is sometimes conducted with the intent to reduce DOC in drainage ditches. In our study, rewetting had the opposite effect and instead increased DOC concentrations and export. Although not explicitly explored and instead based on previous findings from streams and ditches in the area (e.g., Laudon et al., 2011), the particulate contribution to runoff organic C (i.e., POC) is assumed to remain low also after rewetting. We believe this assumption is valid across nutrient-poor boreal peatlands with a similar drainage history, but any effect on POC might deserve further attention when rewetting is conducted in other types of systems. Furthermore, it should be clearly noted that we present only the initial effects (<2 years) following rewetting and that desirable climate benefits or potential effects on water quality may take decades or even centuries to achieve (Ojanen & Minkkinen, 2020). However, the short-term effects that we present here are important to consider when assessing the overall impact of the rewetting. It also stresses the need for long-term monitoring in order to fully understand the biogeochemical consequences of rewetting, and how these consequences might change over time.

Our study revealed important site-specific differences that were ascribable to the different characteristics of the catchments and the outcomes of the rewetting. The open-water pond in one of the catchments which grew in size after rewetting was likely an important driver for increased DIC and CH_4 concentrations and consequently export rates. In addition, the altered runoff C chemistry following rewetting was likely an effect of changes toward longer flow paths through peat and potentially via different soil strata. Based on these results, we believe it will be crucial in future efforts to identify the right locations and methods for rewetting to avoid unwanted effects on water quality and GHG balance (e.g., considering where in the landscape the new ditch initiation point will be formed).

Finally, the studied peatland catchments (both control and rewetted) exported predominantly contemporary, post-bomb, DOC suggesting stable C stores of older peat material. Although there was an indication of a rewetting effect with even younger DOC (^{14}C-DOC enrichment) being exported via runoff at one of the sites following rewetting, the change in ^{14}C content was relatively small and natural variations cannot be excluded. More studies are needed to further explore how the age of the LCE might change following rewetting. Such studies should also include analysis of ^{14}C content in DIC and CH_4 to provide a complete age characterization of the LCE.

These findings underscore the complexity of biogeochemical responses to rewetting and highlight the need for site-specific assessments to effectively manage the C sequestration capacity of peatlands. Given the large interest and increasing plans for rewetting drained peatlands in the near future, the outcomes of this study could be used as a knowledge basis of potential effects on the LCE when implementing such operations.

Conflict of Interest

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

Data Availability Statement

Data used in the current study is accessible on Zenodo: https://doi.org/10.5281/zenodo.14637549 (Zannella et al., 2025).

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