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Tree Harvest Decisions Modulate the Climate Impact of Rewetting in a Low-Productive Peatland Forest in Boreal Sweden

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ABSTRACT

Over the past century, extensive areas of northern peatlands have been drained for forestry. Today, concerns about their role as significant sources of greenhouse gases (GHG) have sparked growing interest in peatland rewetting as a climate mitigating strategy. However, empirical evidence for rewetting effects on ecosystem carbon (C) and GHG balances is still limited, particularly for minerogenic boreal peatland forests. Rewetting of peatland forests also involves decisions about tree harvest, which can have important but understudied consequences for the C cycle. In this study, we quantified tree growth and estimated carbon dioxide (CO_2) and methane (CH_4) fluxes in both peatland areas and ditches over 2 years before (2019–2020) and after (2021–2022) rewetting a low-productive, minerogenic peatland forest in boreal Sweden. We also assessed effects of tree removal during rewetting by comparing harvest and non-harvest areas. Our results suggest that the peatland forest was, on average, C-neutral at the ecosystem-scale during the drained years. After rewetting, the harvested area became a C source (79 g C m^{-2} $year^{-1}$), while the treed area acted as a small C sink (–24 g C m^{-2} $year^{-1}$), with the difference due to diverging responses in net CO_2 exchange. Furthermore, CH_4 emissions doubled after rewetting, resulting in a two- to threefold increase in total GHG emissions (expressed in CO_2 equivalents) over both 20- and 100-year timeframes. While ditches functioned as significant CO_2 sinks and moderate CH_4 sources during the drained years, they became CO_2-neutral and CH_4 emission hotspots after being infilled. Altogether, our findings suggest that rewetting low-productive boreal peatland forests may have a negative short-term climate impact. However, rewetting without tree harvest considerably meliorates ecosystem C and GHG balances. Overall, our study highlights the importance of tree harvesting decisions and the need for a deeper understanding of rewetting as a climate mitigation strategy.

1 | Introduction

Northern peatlands are globally important carbon (C) stores and influence the climate through their exchange of the two most important greenhouse gases (GHGs), that is, carbon dioxide (CO_2) and methane (CH_4), with the atmosphere (Korhola et al. 2010; Loisel et al. 2014). Over the past century, however, extensive areas of natural peatlands have been drained to facilitate timber and crop production (Joosten and Clarke 2002; Minkkinen et al. 2023). Today, these drained areas might act as considerable sources of GHGs (Frolking et al. 2011; Jauhiainen et al. 2019), prompting rapidly growing interest in rewetting programs aimed at mitigating GHG emissions from degraded peatland ecosystems (Günther et al. 2020;

Leifeld and Menichetti 2018). Despite this interest, detailed understanding of short- and long-term effects of rewetting on peatland biogeochemistry and associated GHG balance is currently lacking (Laudon et al. 2025). Thus, developing an empirical knowledge base is critical to support and guide climate change mitigation strategies through rewetting of drained peatlands.

At present, the evaluation of climate benefits from rewetting drained peatlands largely relies on emissions factors (EFs) developed by the IPCC (Hiraishi et al. 2014) and later refined by Wilson et al. (2016). At that time, however, empirical data from rewetted peatlands were limited, and the EFs were primarily derived from data obtained in natural mires, assuming similarities in their respective C cycles. Given that natural mires commonly act as C sinks, EF-based assessments, such as those used in policy contexts (e.g., Jordbruksverket 2018; UNFCCC 2024), often predict immediate climate benefits from rewetting, suggesting a rapid transition from a C source to a C sink within the first year after rewetting. However, functional differences (hydrology, geochemistry and vegetation) between natural and rewetted peatlands may persist for at least three decades (Kalhori et al. 2024). Studies from rewetted temperate grassland (Kreyling et al. 2021; Peacock et al. 2019; Renou-Wilson et al. 2016) and boreal peat extraction areas (Nugent et al. 2018; Tuittila et al. 1999; Yli-Petäys et al. 2007) suggest that the establishment of a C sink may be delayed for several years. Interestingly, a rewetted boreal fen was recently reported to remain a net C and GHG source during the first 3 years following rewetting (Tong et al. 2025). Despite these findings, studies that directly compare GHG balances before and after rewetting are still scarce, particularly for boreal minerogenic mires that were drained for forestry over a century ago.

The primary aim of peatland rewetting as a climate mitigation strategy is to reduce peat decomposition and associated CO_2 emissions by raising the water table level and creating anoxic conditions (Hiraishi et al. 2014; Leifeld and Menichetti 2018). However, the climate benefit from reduced soil CO_2 emissions might be counteracted by increased CH_4 emissions, which are promoted under anaerobic conditions established by rewetting (Harpenslager et al. 2015; Koskinen et al. 2016; Vanselow-Algan et al. 2015). In particular, drainage ditches have been suggested to act as CH_4 emission hotspots following ditch-blocking, with rates up to an order of magnitude higher than those from the surrounding peatland area (Cooper et al. 2014; Waddington and Day 2007). Considering the substantially higher global warming potential of CH_4, approximately 80 and 27 times higher than that of CO_2 over 20- and 100-year timeframes, respectively, it may take several decades to achieve a net climate benefit (i.e., cooling effect) when accounting for the radiative forcing dynamics of both gases following rewetting of drained boreal peatlands (Günther et al. 2020; Launiainen et al. 2025; Ojanen and Minkkinen 2020). Therefore, detailed knowledge on the trade-offs between reduced CO_2 and increased CH_4 emissions under site-specific conditions is needed to adequately develop and evaluate rewetting strategies aimed at climate change mitigation.

When rewetting drained forested peatlands, a key management decision is whether to harvest and remove the existing trees prior to rewetting (i.e., restoring toward an open, treeless peatland) or to leave them standing (i.e., restoring toward a tree-covered peatland). On the one hand, tree harvest may create economic revenue for landowners and reduce water loss via transpiration, with the latter supporting the rewetting process. In addition, harvested wood may provide climate benefits by substituting fossil fuel emissions in the industry sector (Launiainen et al. 2025; Ojanen and Minkkinen 2020). On the other hand, the economic value of harvesting, particularly that of low-productive stands in the boreal region, might be limited. In such cases, retaining the tree stand for continued timber production, along with associated C sequestration and potential biodiversity benefits after rewetting, may be a more viable strategy (Laine et al. 2024; Makrickas et al. 2023; Peura et al. 2018). Given the C uptake provided by the tree layer, the decision to harvest or retain trees has potential implications for rewetting effects on the ecosystem-scale C and GHG balances (Laine et al. 2024; Launiainen et al. 2025; Ojanen and Minkkinen 2020). Additional uncertainty arises from the lack of knowledge on tree growth responses to rewetting, which may range from continued growth to gradual or instant die-off (Edvardsson et al. 2024; Edvardsson and Hansson 2015). Thus, the consequences of stand management for the short- and long-term climate impact of rewetting remain uncertain.

In this study, we combined chamber-based measurements of CO_2 and CH_4 fluxes from both the peatland area and drainage ditches with tree growth analysis to assess the initial effects of rewetting on the ecosystem C and GHG balances of a low-productive boreal forested peatland in northern Sweden. Measurements were conducted over 2 years before (2019–2020) and after (2021–2022) rewetting. The main objectives were to: (i) determine the effects of rewetting on individual components of the ecosystem C and GHG balances (i.e., CO_2 and CH_4 fluxes from/into soil, trees, ground vegetation, and ditches), and (ii) explore the effects of tree harvest during rewetting on the overall ecosystem C and GHG balances (Figure 1).

2 | Materials and Methods

2.1 | Site Description

The study was carried out at the peatland rewetting site within the Trollberget Experimental Area (TEA; Laudon et al. 2023), located in Västerbotten County, northern Sweden (64°10′15.61″ N, 19°50′14.06″ E; Figure 2). The region has a continental subarctic climate (Dfc based on Köppen classification) with a 30-year (1991–2020) mean annual air temperature (Ta) and precipitation (PPT) of 3.0°C and 635 mm, respectively. The snow-free season typically extends from around mid-May to mid-November. Peat depth at the site averages 2.4 m, ranging from 0.2 to over 6 m ($n = 190$). The mean C:N ratio in the upper 0–20 cm peat layer is 43 (Casselgård 2020).

The study site was originally a minerogenic, oligotrophic mire with characteristics of a flark fen that was drained around 1910 (Tong et al. 2025; Nordstedt et al. 2021). Drainage was carried out manually, resulting in a sparse ditch network consisting of a main ditch (about 5 m wide and 2 m deep) running east–west through the center of the site, along with a few smaller auxiliary ditches (about 1 m wide and 1 m deep). The ditch network was not maintained in the years following drainage, leading to a decline

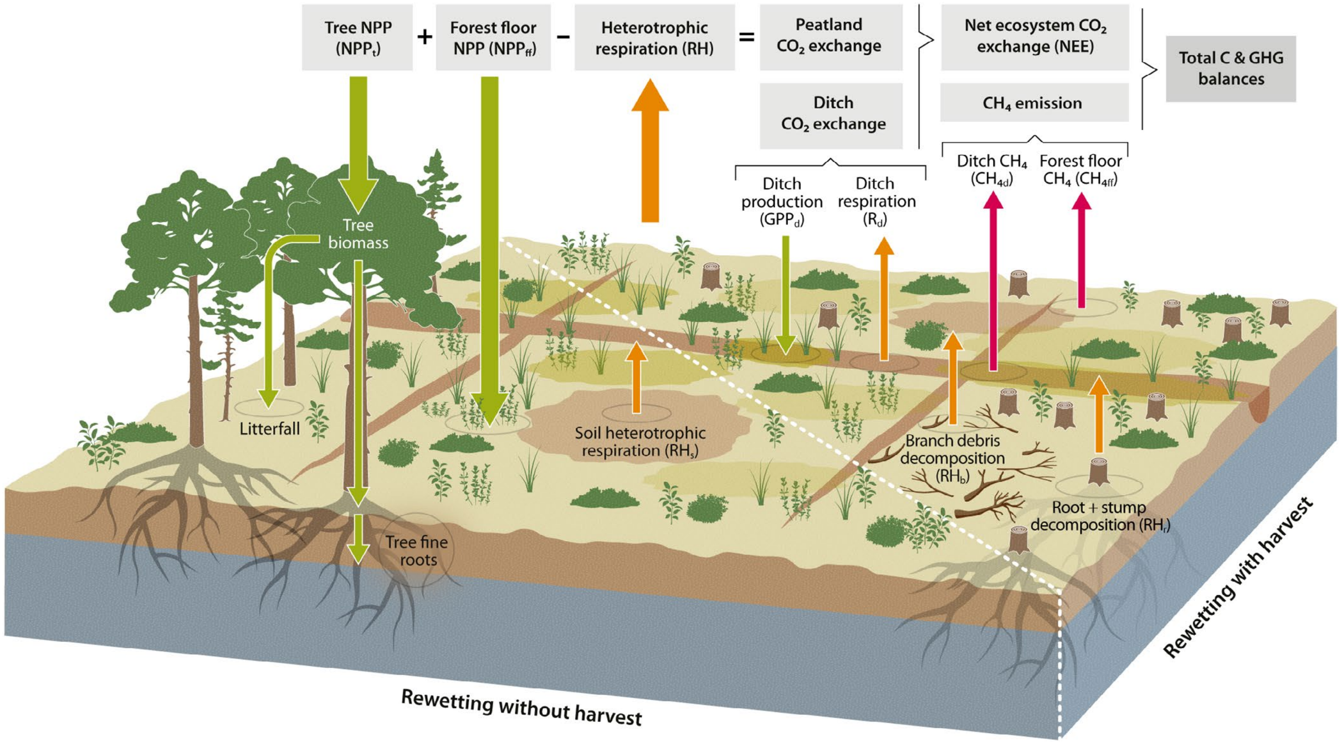

FIGURE 1 | Conceptual diagram illustrating the carbon dioxide (CO_2) and methane (CH_4) fluxes and their total carbon (C) and greenhouse gas (GHG) balances determined in this study across the rewetted peatland forest area including a section with and without tree harvest during rewetting (separated by the white dotted line). Fluxes include annual tree net primary production (NPP_t; composed of changes in tree biomass, litterfall and fine root production), forest floor vegetation NPP_{ff}, heterotrophic respiration (RH; composed of heterotrophic respiration from soil [RH_s], branch + foliage debris [RH_b], and root + stumps [RH_r]), ditch gross primary production (GPP_d) and respiration (R_d), as well as methane emissions from the forest floor (CH_{4ff}) and ditches (CH_{4d}).

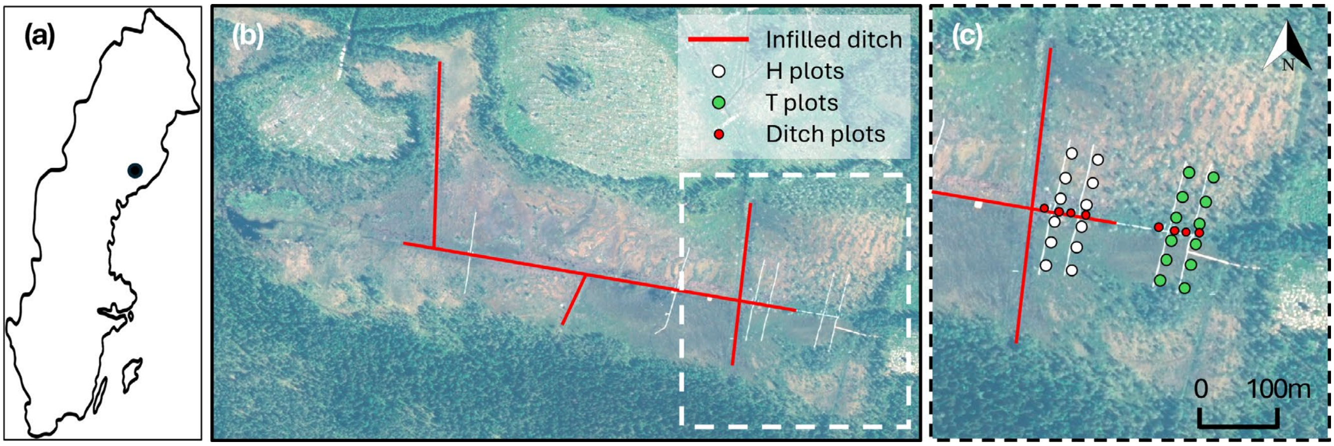

FIGURE 2 | (a) Location of the study site (Trollberget rewetted peatland) in northern Sweden, (b) satellite image of the entire site (before rewetting), with red lines illustrating the location of the infilled drainage ditches, and (c) zoomed-in image of the study measurement area, including sampling plots for the Harvested (H; white circles) and Treed (T; green circles) treatment areas. Red circles indicate manual chamber flux measurement locations inside of the main ditch.

in drainage function due to peat infilling and subsequent vegetation ingrowth (including floating *Sphagnum* mats and vascular plants, primarily *Carex* spp.). Compared with active peatland drainage programs initiated in other Nordic countries in recent decades, particularly in Finland, the relatively sparse ditch density and lack of maintenance are typical of drainage networks established over a century ago in boreal Sweden.

Following drainage, the treed areas (primarily located in the northern and eastern parts of the site) developed into a low-productive ($<1\,m^3\,ha^{-1}\,year^{-1}$ aboveground wood volume increment) peatland forest dominated by Scots pine (*Pinus sylvestris* L.; basal area $= 2.6\,m^2\,ha^{-1}$). At the time of the study, the vegetation was dominated by the residual plant community remaining following drainage. Specifically, this included vascular plants, mainly shrubs, such as bog bilberry (*Vaccinium uliginosum* L.), bog cranberry (*V. oxycoccos* L.), black crowberry (*Empetrum nigrum* L.), common heather (*Calluna vulgaris* L. (Hull)) and bog rosemary (*Andromeda polifolia* L.); mosses, particularly *Sphagnum* spp.; and graminoids, such as cottongrass (*Eriophorum vaginatum* L.). Vegetation in the drainage ditches included *Sphagnum* spp. mats, *Carex rostrata* (Stokes) and *C. lasiocarpa* (Ehrh.) during the drained years, while rapid colonization with *E. vaginatum* occurred following ditch-blocking.

The site was rewetted in November 2020 using a 20-ton crawling excavator, following conventional authority-defined methods. The rewetting activities involved removing the sparse tree cover and infilling of the ditches with on-site peat and harvested tree logs across ~90% of the area. The biomass of trees and stumps was left on site. A small section (~10%) was initially preserved as a control, with no tree harvest or ditch-infilling activities, to enable a Before-After-Control-Impact (BACI) experimental design. However, spontaneous rewetting via lateral groundwater inflow from the adjacent rewetting area raised the water level in this section to a similar level ($p > 0.05$). Consequently, this area was finally included as part of the rewetted treatment.

2.2 | Experimental Study Design

In this study, all measurements were carried out in the eastern part of the site, where post-drainage development led to the formation of a low-productive peatland forest (Figure 2c). The experimental design included two treatments: a Harvested treatment (H), where the trees were removed as part of the rewetting activities, and a Treed treatment (T), where the trees were left standing. In each treatment area, two parallel boardwalk transects were established perpendicular to the main ditch, extending 50 m north and south from the ditch. On both sides, sampling locations were established 5, 25 and 50 m away from the ditch, resulting in 12 locations for each of the treatment areas (i.e., $n = 12$; Figure 2c). At each location, a paired setup was used to measure fluxes from both the vegetated forest floor and from bare peat (i.e., vegetation free) surfaces. For the latter, 1 m^2 areas were established in autumn 2018 by clipping all green vegetation, including the green moss parts. These areas were kept free of vegetation throughout the study years. To exclude root respiration, roots were trenched annually to a depth of ~30 cm along the edges of these vegetation removal areas using a sharp knife.

2.3 | Chamber Measurements of CO_2 and CH_4 Fluxes

2.3.1 | Forest Floor Fluxes

At each sampling location, CO_2 and CH_4 fluxes were quantified using the closed dynamic chamber method at biweekly intervals from mid-May to late October (i.e., during frost-free conditions) in each of the 2 years before (2019–2020) and after (2021–2022) rewetting. For this purpose, square aluminum frames (45 × 45 cm) with a groove for air-tight sealing were permanently installed to a soil depth of 12 cm in both the natural and vegetation removal plots. Forest floor net CO_2 exchange (NE_{ff}) and CH_4 exchange (CH_{4ff}) were measured in the natural plots using a transparent Plexiglas chamber (92% transparency; 20 cm height, 40 L volume) connected via 6 mm (outer ⌀) tubing to a portable CO_2, CH_4 and H_2O gas concentration analyzer (GasScouter G4301; Picarro Inc., Santa Clara, CA, USA). The chamber was placed onto the frame for a 3-min deployment period, during which gas concentrations were recorded at ~1.5 s intervals. Return air flow from the analyzer into the chamber provided constant air mixing within the chamber headspace. The chamber was equipped with sensors for photosynthetically active radiation (PAR) and Ta recorded every 5 s (HOBO Pendant MX2202; Onset Comp., Bourne, MA, USA).

After each transparent chamber measurement, forest floor respiration (R_{ff}) was determined from a subsequent measurement during which the chamber was covered with an opaque tarp. Heterotrophic respiration from bare soil (RH_s) was measured in the vegetation removal plots using the same setup as for R_{ff}. Gross primary production of the forest floor vegetation (GPP_{ff}) was derived from the difference between the measured NE_{ff} and R_{ff} fluxes (i.e., $GPP_{ff} = NE_{ff} - R_{ff}$). Next, net primary production of the forest floor vegetation (NPP_{ff}) was estimated from GPP_{ff} assuming a carbon use efficiency (CUE) of 0.5 (i.e., $NPP_{ff} = GPP_{ff} \times CUE$) (Jin et al. 2025). This approach was chosen because, although ecosystem-scale NPP can be derived as the difference between net ecosystem CO_2 exchange (NEE) and total heterotrophic respiration (RH; including respiration from soil, decaying roots, and slash debris; Luyssaert et al. 2007), the NE_{ff} measurement in this study included tree root respiration but not tree root production in the drained years and in the rewetted T treatment, which violates the mass-balance assumption required for partitioning NE_{ff}. Throughout the frost-free season, flux measurements were carried out in a randomized plot order to avoid diurnal effects. In this study, all fluxes are expressed following the atmospheric sign convention, where positive fluxes represent emissions to the atmosphere and negative fluxes indicate uptake.

2.3.2 | Ditch Fluxes

We also measured CO_2 and CH_4 fluxes from the main drainage ditch at a bi-weekly interval from mid-May to late October during 2020–2022. For this purpose, two aluminum frames were permanently installed at the bottom of the ditch close to each side of the four boardwalk transect crossings over the ditch (Figure 2c). Measurements of ditch net CO_2 exchange (NE_d), total respiration (R_d), gross primary production (GPP_d), and CH_4 fluxes (CH_{4d}) were conducted using the same chamber and measurement protocol as described above for forest floor fluxes. However, a larger chamber (50 cm height, 100 L volume) was used at certain locations and during the peak growing season when ditch vegetation was tall. This large chamber was equipped with a low-speed fan to support constant air mixing within the chamber headspace, as well as sensors to measure PAR and Ta (HOBO Pendant MX2202; Onset Comp., Bourne, MA, USA). During wet periods, when the ditch was filled with water and the permanent flux frames were inaccessible, a polyethylene foam ring was attached to the base of either the small or large chamber (depending on vegetation height) to enable floating chamber measurements on the water surface. In the rewetting years, CO_2 and CH_4 fluxes at the T treatment, where the ditch was not filled, were assumed to be equal to those from the H treatment for subsequent analysis.

2.3.3 | Flux Calculation and Quality Control

To estimate CO_2 and CH_4 fluxes, we first calculated the linear change in gas concentrations within the chamber headspace over time. Multiple linear slopes for concentration changes were calculated using a moving window of 40 data points (~1 min), incremented by one-point across the entire 3-min measurement period. The first and last 5 s of each measurement were excluded to avoid artifacts from chamber placement and removal (i.e., a start and end "dead band"). This stepwise moving window approach was employed to automatically discard short periods during chamber deployment when fluctuating PAR levels during partly cloudy days disturbed parts of the transparent chamber measurement. The slope of the window with the highest coefficient of determination (R^2) was selected as the final slope. To estimate the flux rate, this slope was then adjusted by the frame area, the volume of the chamber headspace, air pressure and molar mass of gas at given chamber Ta.

Final quality control procedure included the removal of poor-quality fluxes using a combination of root mean square error (RMSE) and R^2 as goodness of fit measures. Specifically, CO_2 fluxes with RMSE $>0.50\,\mu\text{mol CO}_2\ \text{m}^{-2}\,\text{s}^{-1}$ and $R^2<0.90$ ($p<0.001$), and CH_4 fluxes with RMSE $>0.02\,\mu\text{mol CH}_4\ \text{m}^{-2}\,\text{s}^{-1}$ and $R^2<0.90$ ($p<0.001$) were discarded. These thresholds were determined by examining the relationships among flux magnitudes, RMSE, and R^2. Sporadic observations (< 1% of all records) of negative R_{ff} and RH_s fluxes (i.e., apparent uptake due to disturbance) were also removed.

2.4 | Tree Net Primary Production

Annual tree net primary production (NPP_t) was estimated as the sum of tree aboveground and coarse root (⌀ > 1 cm) biomass production (Section 2.4.1), fine root (⌀ ≤ 2 mm) biomass production (Section 2.4.2) as well as litterfall production (Section 2.4.3). For all components, annual biomass production was converted to C units assuming a dry biomass C content of 50%.

2.4.1 | Tree Aboveground and Coarse Root Biomass Production

At each flux sampling location, a tree inventory plot was established for tree biometric measurements: a 5 m radius plot at the 5 m distance, and 10 m radius plots at the 25 and 50 m distances from the ditch. Within each of the 24 inventory plots, species, diameter at breast height (DBH; at 1.3 m), and height of all living and dead standing trees were recorded in late September 2020, prior to rewetting. The same procedure was repeated for the 12 plots in the T treatment in June 2024. Saplings (DBH < 5 cm) were not included in the inventory as their total biomass and production were considered negligible. Both tree aboveground and coarse root biomass stocks were estimated as a function of tree DBH and height using species-specific allometric equations developed for peatland forests (Finér 1989; Minkkinen et al. 2001).

Increment cores were collected from selected trees in late September 2020 and June 2024 using a 5.15 mm Ø increment borer (Haglöf Sweden AB, Långsele, Sweden). Two increment cores per tree were collected from 45 representative trees in each of the H and T treatments in 2020, and from the same 45 trees at the T treatment in 2024. Trees were selected to represent all DBH classes (in 5 cm intervals) with the number of samples per class proportional to the number of trees in each class. In the laboratory, all cores were air-dried, mounted on wooden holders, and sanded with fine sandpaper until annual rings were clearly visible. Cores were scanned using a high-resolution scanner, and ring widths were measured using WinDENDRO (Regent Instruments Inc.) in 2020 and CooRecorder 9.8.1 (Cybis Elektronik & Data AB) in 2024. From the ring-width series, we reconstructed annual DBH for all trees within each inventory plot for the period 2019–2022.

Annual tree production was then estimated by combining tree biomass stock data with annual radial growth rates derived from tree cores. Specifically, radial increments were used to adjust the 2020 and 2024 biomass stock estimates, allowing calculation of annual biomass changes for the periods 2019–2020 and 2021–2022, respectively.

2.4.2 | Tree Fine Root Production

Annual tree fine root production was estimated using an established empirical model developed for boreal peatland forests (Lehtonen et al. 2016). The model predicts fine root biomass based on long-term mean temperature sum and stand basal area, the latter obtained from the forest inventory data. Annual tree fine root production was then estimated as the product of annual fine root biomass and a turnover rate, the latter assumed to be 1.2 following previous studies (Samariks et al. 2024).

2.4.3 | Litterfall

Foliage biomass production was estimated using litterfall traps (0.25 m^2) placed at the centre of each inventory plot ($n=12$ per treatment for both H and T). The traps were emptied in May to capture winter litter, once in September to capture summer litter, and then monthly from September to November to capture the peak litterfall during autumn. The collected material was sorted into three categories: foliage (pine needles), woody twigs, and other organic material (containing seeds and cones). Samples were oven dried (60°C, 48 h) to determine their dry weight.

2.5 | Meteorological, Soil Environmental and Vegetation Measurements

During each flux sampling campaign, ambient Ta was recorded using a shaded temperature sensor mounted outside of the small and large chambers (HOBO Pendant MX2201; Onset Comp., Bourne, MA, USA). Meanwhile, ambient PAR was measured in a nearby open area free from any shading obstacles using a light and temperature logger (HOBO Pendant MX2202; Onset Comp., Bourne, MA, USA). It is noteworthy that shading from the sparse tree canopy did not significantly reduce PAR below the canopy relative to ambient (i.e., above-canopy) PAR (Figure S1). In addition, soil temperature (Ts) was recorded adjacent each flux frame at 5 and 10 cm depths (Ts5 and Ts10) using a handheld digital thermometer (M514B, Sunartis, Mingle Instruments GmbH Europe, North Rhine-Westphalia, Germany). Water table level (WTL) relative to the ground surface was manually recorded during each flux campaign using perforated PVC tubes installed near each flux frame. At ditch locations, temperature measurements reflected either soil or water temperature depending on whether the ditch was dry or water-filled, respectively. Water depth was recorded near each flux frame with a meter stick during inundated periods when water was present in the ditch.

In each treatment area (H and T), automatic water level loggers (Levelogger 5 Junior; Solinst Canada Ltd., Georgetown, ON, Canada) were installed in October 2019 along one of the two replicate transects at each of the six sampling plots to obtain continuous WTL data at 30-min intervals. These WTL records were calibrated against manual WTL measurements taken at the same locations. In addition, temperature loggers (HOBO TidbiT MX2204; Onset Comp., Bourne, MA, USA) were installed in August 2019 at a 5 cm depth along the same transects to continuously monitor Ts. For the remaining transects, continuous WTL data were derived by correlating local manual WTL data with those from the automated loggers installed at the adjacent transect.

Since continuous on-site data for key meteorological variables were not available prior to November 2020 (PAR and Ta), August 2019 (Ts), and October 2019 (WTL), we used half-hourly resolution data from nearby ICOS stations. Specifically, PAR and Ta were obtained from the ICOS SE-Svb forest station located 7.8 km from the study site (Peichl et al. 2025), while Ts and WTL data were sourced from the ICOS SE-Deg peatland station, 13.6 km away (Nilsson et al. 2025). These data were correlated with on-site data once available, allowing us to reconstruct continuous time series from the beginning of 2019. These continuous time series were essential as input drivers for the semi-empirical models used to extrapolate the chamber fluxes to the annual scale (see Section 2.6.2).

To monitor vegetation development at high temporal resolution, a greenness index was derived from hourly images collected through digital repeat photography. Images were taken with a battery-operated outdoor camera (TimelapseCam 8.0, model WSCT01; Wingscapes, Calera, AL, USA) mounted on a vertical pole at 3 m above the ground surface with a downward looking angle of 15°, viewing a representative section of the study area. Vegetation greenness index was calculated within a selected region of interest based on the green chromatic coordinate (gcc) as described in Peichl et al. (2015) (Equation 1):

$$\text{gcc} = G / (R + G + B) \quad (1)$$

where R, G, and B are the digital numbers (0–255) of the red, green, and blue image channels, respectively.

To estimate frame-specific gcc, overhead images of each vegetated flux frame along the transects were taken using a tablet computer (Samsung Galaxy Tab, Samsung Electronics Co. Ltd., Suwon, South Korea) during each flux sampling campaign. The continuous site-mean gcc site series was then multiplied with the relative differences in gcc among frames to obtain continuous, frame-specific gcc estimates for the growing season. These values were used as input for the semi-empirical models developed for extrapolating the chamber flux data to the annual scale (see Section 2.6.2).

2.6 | Annual C and GHG Balance of Peatland Area and Ditches for Harvested and Treed Treatments

The annual total ecosystem C balance was calculated from the sum of net ecosystem CO_2 exchange (NEE) and CH_4 exchange from the peatland area and its ditches (Equation 2) for the H and T treatments (see also Figure 1).

$$\text{Total C balance} = \text{NEE} + CH_4 \quad (2)$$

For the peatland area, the annual net CO_2 exchange (NE_p) was derived as the difference between NPP and total heterotrophic respiration (RH; Equation 3), where NPP is the sum of NPP_t and NPP_{ff} (Equation 4). In most cases, RH was equal to the heterotrophic respiration from bare soil (RH_s). However, for the H treatment during the rewetted years, RH also encompassed estimates for respiration losses from decomposing roots and stumps (RH_r) and branch debris (RH_b; including foliage) (Equation 5). For the ditches, annual net CO_2 exchange (NE_d) was calculated as the difference between annual ditch GPP and respiration (GPP_d and R_d; Equation 6).

$$NE_p = NPP + RH \quad (3)$$

$$NPP = NPP_t + NPP_{ff} \quad (4)$$

$$RH = RH_s + RH_r + RH_b \quad (5)$$

$$NE_d = GPP_d + R_d \quad (6)$$

Annual sums of NPP_{ff}, RH_s, GPP_d, R_d and CH_4 fluxes were estimated by extrapolating manual chamber measurement using semi-empirical models as described below (Sections 2.6.1 and 2.6.2). To determine RH_r and RH_b, we used tree inventory data from September 2020 and species-specific allometric equations from Minkkinen et al. (2001) to estimate the biomass of roots, stumps, and branches (including foliage) of the standing trees in the H treatment area. Species-specific decay functions obtained from the literature (Shorohova et al. 2008; Yatskov et al. 2003) were then applied, assuming constant rates under various environmental conditions and a biomass C content of 50%, to obtain the annual C losses via RH_r and RH_b following harvest.

2.6.1 | Modeling of Annual CO_2 and CH_4 Fluxes in the Peatland Area

We developed semi-empirical models using measured fluxes and concurrent environmental data to upscale chamber-based

CO_2 and CH_4 fluxes to the annual scale. Separate models were parameterized for each year and each treatment (aggregated across all measurement locations) and were further separated for growing season and non-growing season periods, with the latter defined as periods when $Ts5 < 2°C$. Continuous environmental driver data were used to apply these models across the full year. All model parameters and performance statistics are provided in Tables S1–S3.

We modeled RH_s (mg CO_2-C $m^{-2} h^{-1}$) using the exponential model presented by Lloyd and Taylor (1994) (Equation 7):

$$RH_s = R_{10}\ \exp^{E_0\left(\frac{1}{56.02} - \frac{1}{Ts5 - 227.13}\right)} \qquad (7)$$

where Ts5 is soil temperature (K) at 5 cm depth, R_{10} represents base respiration at 10°C and E_0 represents the activation energy parameter (K). Using this model, hourly RH_s was then predicted based on the hourly Ts5 data. While the growing season models were based on all measured data, only the measurements taken in May and October (during which soils and plants were close to winter dormancy conditions) were used to obtain the non-growing season model parameters. It is noteworthy that the E_0 parameters of the non-growing season models were significantly lower compared to the growing season models.

Annual GPP_{ff} (mg CO_2-C $m^{-2} h^{-1}$) was predicted based on PAR and vegetation development. Specifically, GPP_{ff} fluxes from each vegetated sample plot were fitted to PAR inside the chamber using a hyperbolic function modified by the frame-specific gcc (Equation 8):

$$GPP_{ff} = \frac{\alpha \times A_{max} \times PAR \times gcc_{norm}}{(\alpha \times PAR) + \left(A_{max} \times gcc_{norm}\right)} \qquad (8)$$

where PAR is the photosynthetically active radiation ($\mu mol\ m^{-2} s^{-1}$) inside the chamber, α is the light-use efficiency of photosynthesis (i.e., the initial slope of the light response curve, mg CO_2-C μmol photons^{-1}), A_{max} is maximum photosynthesis at light saturation (mg CO_2-C $m^{-2} h^{-1}$), and gcc_{norm} is the frame-specific greenness index (i.e., green chromatic coordinate) normalized to scale between 0 and 1. Using this model, hourly GPP_{ff} was then extrapolated during the growing season based on the continuous PAR and frame-specific gcc_{norm}. During the non-growing season period, GPP_{ff} was assumed to be zero. Annual NPP_{ff} was finally derived from multiplying GPP_{ff} with a CUE of 0.5.

Forest floor CH_4 fluxes (CH_{4ff}; mg CH_4-C $m^{-2} h^{-1}$) were extrapolated at hourly resolution with a semi-empirical model based on soil temperature and WTL (Equation 9):

$$CH_4 = \exp^{b_0 + b_1 \times Ts10 + b_2 \times WTL} \qquad (9)$$

where Ts10 is soil temperature (°C) at 10 cm depth and WTL is water table level (cm). Here, Ts10 rather than Ts5 was chosen since it was more strongly correlated to the CH_4 flux. As no significant non-growing season model could be established, the 10th percentile of October CH_4 fluxes was used as a constant minimum flux during this period.

2.6.2 | Modelling of Annual CO_2 and CH_4 Fluxes in the Drainage Ditch

Ditch respiration (R_d) during the growing season was modelled using the same Equation (7), based on temperature. Since no significant model could be established for the non-growing season, the 10th percentile of October R_d fluxes was extrapolated throughout this period. We then modelled GPP_d based on PAR and vegetation development using a hyperbolic function as in Equation (8). Ditch vegetation phenology was assumed to follow the mean seasonal trajectory of the peatland forest-floor vegetation, and the mean gcc_{norm} from the transects was used. For CH_{4d}, no significant models could be established with any of the available environmental drivers (Ts, WTL). Therefore, we simply extrapolated the median values of all measurements for the growing season, and the 10th percentile of the October fluxes was applied to the non-growing season period. We note that in the rewetting years 2021–2022, annual CO_2 and CH_4 balances at the T treatment (where ditches were not filled) were assumed to be equal to those from the H treatment.

2.6.3 | Area-Weighted C and GHG Balances for Harvested and Treed Treatments

We determined the relative contribution of the peatland area (96.5%) and ditches (3.5%) to the total area (with the same contributions assumed for both H and T treatments) to compute the area-weighted ecosystem C and GHG balances. This approach allowed us to evaluate the proportional influence of these two distinct ecosystem components to the total balances. For the ditches, we assumed that C and GHG fluxes measured in the main ditch were representative for those in the side ditches. To calculate the GHG balance, fluxes were converted to CO_2 equivalents (CO_2-eq), applying global warming potentials (GWPs) of 81 and 27 for CH_4 over 20 and 100-year timeframes, respectively (Lee et al. 2023).

2.7 | Statistics

Since the flux data was based on repeated-measures (i.e., not independent) and not normally distributed, the non-parametric Wilcoxon rank sum test was used to assess significant differences between treatment medians. Specifically, we compared the H and T treatments for differences in GPP, NPP, RH, Ra and CH_4 fluxes and biomass pools. Similarly, the test was also applied to compare drained and rewetted periods for significant effects. Statistical significance level was $p < 0.05$ unless stated otherwise. All calculations and statistics were computed using MATLAB software (MATLAB R2024b, Mathworks, Natick, MA, USA).

3 | Results

3.1 | Environmental Conditions During the Study Period

During the study period 2019–2022, annual mean Ta was 2.3°C, 4.1°C, 2.2°C, and 3.2°C, respectively. These included one warmer and one normal year in both the drained and

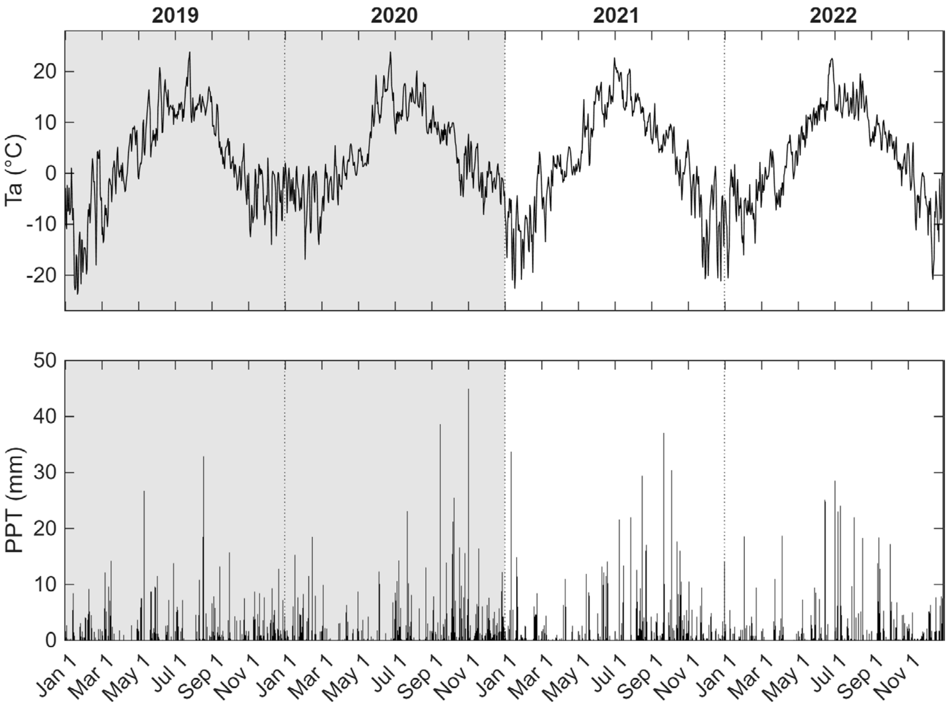

FIGURE 3 | Daily mean air temperature (Ta, top panel) and precipitation sum (PPT, bottom panel) during the drained (2019–2020; shaded area) and rewetted (2021–2022) years. Data were obtained from the Svartberget Climate Station "Hygget", located approximately 7 km from the study site.

rewetted periods relative to the 30-year (1991–2020) long-term mean of 2.4°C (Svartberget Climate Station "Hygget"). The growing season (May 15 to October 15) mean Ta was 10.7°C in 2019 and 11.7°C in each of the years 2020–2022. Seasonal patterns of daily mean Ta were generally similar across growing seasons, except for an exceptional early warm summer period in June 2020, with a mean of 16.3°C, relative to 13.5°C to 14.4°C in the other years (Figure 3a). Annual precipitation was 617, 795, 766 and 639 mm in 2019–2022, respectively. These included 1 year with precipitation close to the 30-years mean (635 mm) and one wetter year in both the drained and rewetted periods. It is noteworthy that precipitation during July and August of the drained years 2019 and 2020 was considerably lower (113 and 136 mm) compared to the same period in the rewetted years 2021–2022, which received 190 and 175 mm, respectively (Figure 3b).

Averaging bi-weekly manual environmental measurements taken during flux measurements suggests that the mean growing season WTL for the drained years 2019–2020 was −25 and −21 cm below surface at the H and T treatments, respectively, with minimum WTLs reaching −71 and −66 cm (Figure 4). Following rewetting, the mean growing season WTL averaged over 2021–2022 was −12 cm at both the H and T treatments, with minimum values of −40 and −38 cm, respectively. Growing season mean Ts5 at the H and T treatments was 12.0°C and 11.3°C averaged over the drained years, compared to 13.2°C and 13.0°C during rewetted years, respectively (Figure 4). Maximum greenness (i.e., gcc_{max}) of the forest floor vegetation was 0.34 and 0.36 at the H and T treatments during the drained year 2020, compared to 0.35 and 0.37 averaged over the rewetted years 2021–2022, respectively (Figure 4).

3.2 | Forest Floor CO_2 and CH_4 Fluxes

During the drained years 2019 and 2020, forest floor NPP (i.e., NPP_{ff}) reached maximum rates of −137 and −134 mg C $m^{-2}h^{-1}$ in the H and T treatments, respectively (Figure 5). In the rewetted years 2021 and 2022, maximum NPP_{ff} rates increased to −163 and −183 mg C $m^{-2}h^{-1}$ in the H and T treatments, respectively. When averaged over both years, growing season mean NPP_{ff} rates did not differ significantly between the H and T treatments in either the drained or rewetted years. However, NPP_{ff} at the T treatment was significantly greater (i.e., more negative) during the rewetted period compared to the drained period, while at the H treatment, the difference between the periods was not quite significant ($p = 0.06$).

Soil heterotrophic respiration (RH_s) reached maximum rates of 105 and 107 mg C $m^{-2}h^{-1}$ in the H and T treatments, respectively, during the drained years 2019 and 2020 (Figure 5). While similar maximum rates were observed during the first year after rewetting, maximum RH_s decreased to 81 and 75 mg C $m^{-2}h^{-1}$ in the H and T treatments, respectively, during the second rewetting year (Figure 5). When averaged over the growing season, RH_s did not differ significantly between the H and T treatments, nor between the drained and rewetted years.

During the drained years, CH_4 emissions generally remained below 2.0 mg C $m^{-2}h^{-1}$ at both the H and T treatments, with growing season means significantly higher at the T treatment relative to the H treatment (Figure 5). In the rewetted years, maximum CH_4 emission rates increased to 2.5 and 3.2 mg C $m^{-2}h^{-1}$ in 2021 and 4.3 and 5.4 mg C $m^{-2}h^{-1}$ in 2022 at the H and T treatments, respectively. However, growing season

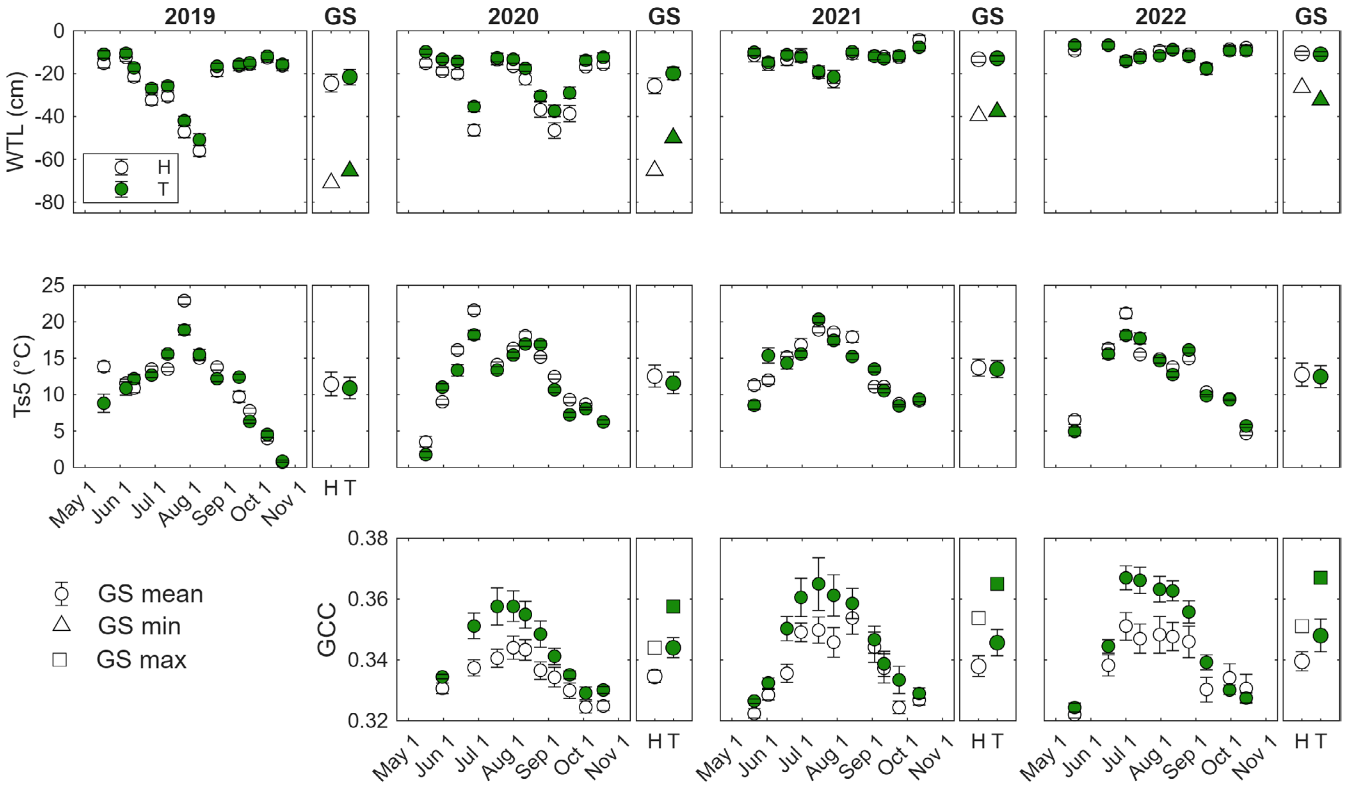

FIGURE 4 | Bi-weekly environmental conditions during flux measurement campaigns, including growing season (GS, May 15 to October 15) means, maxima, and minima for water table level (WTL, top row), soil temperature at 5 cm depth (Ts5, middle row) and greenness index (gcc, bottom row) for the Harvested (H) and Treed (T) treatments over the drained (2019–2020) and rewetted (2021–2022) years. Error bars are showing one standard error. Note that the H transect was treed during 2019–2020 and harvested in 2021–2022.

means no longer differed between the H and T treatments in either year. Compared to the drained years, CH_4 emissions were significantly higher at both H and T treatments following rewetting.

3.3 | Exchanges of CO_2 and CH_4 in Ditches

In the drained year 2020, GPP of ditch vegetation (GPP_d) reached maximum rates of −981 and −800 mg C $m^{-2}h^{-1}$ at the H and T treatments, respectively, with no significant difference in growing season means (Figure 6). In the first year after rewetting, maximum GPP_d at the H treatment was reduced (i.e., less negative) to −188 mg C $m^{-2}h^{-1}$ but increased again in the second rewetting year to −688 mg C $m^{-2}h^{-1}$, with the growing season mean GPP_d being significantly lower only in the first rewetting year relative to the drained year.

Ditch respiration (R_d), including contributions from both vegetation and soil, reached maximum rates of 306 and 297 mg C $m^{-2}h^{-1}$ in the H and T treatments, respectively, in the drained year 2020, with no significant difference in growing season means (Figure 6). In the rewetting years, maximum R_d in the H treatment reached 146 and 214 mg C $m^{-2}h^{-1}$ in 2021 and 2022, respectively, with the growing season mean of the first rewetting year being significantly reduced relative to the drained year.

Methane emissions from the ditch (CH_{4d}) in the drained year 2020 reached maximum rates of 9 and 5 mg C $m^{-2}h^{-1}$ at the H and T treatments, respectively, with no significant difference in growing season means (Figure 6). In the first rewetting year, maximum CH_{4d} emissions at the H treatment were similar to those in the drained year 2020. In the second rewetting year, CH_{4d} emission rates at the H treatment ranged between 19 and 29 mg C $m^{-2}h^{-1}$ for most of the growing season (i.e., early-July to mid-September), with the growing season mean being significantly higher than that in the drained year.

3.4 | Annual C and GHG Balances for Harvested and Treed Treatments

3.4.1 | Peatland Areas

During the drained years, the annual NPP_{ff} ranged from −200 to −192 g C $m^{-2}year^{-1}$ at the H treatment and −205 to −200 g C $m^{-2}year^{-1}$ at the T treatment, contributing approximately 80% of total NPP (Table 1). Following rewetting, NPP_{ff} decreased by about 6% at the H treatment but increased by about 9% in the T treatment. Total tree net primary production (NPP_t), which contributed the remaining 20% of NPP during the drained years, ranged from −47 to −54 g C $m^{-2}year^{-1}$ across both treatments and years. In the rewetted years, NPP_t in the T treatment increased on average by 17%.

Annual RH_s was similar between the H and T treatments during the two drained years but was about 12% higher in 2019 (267 and 280 g C $m^{-2}year^{-1}$) compared to 2020 (242 and 244 g C

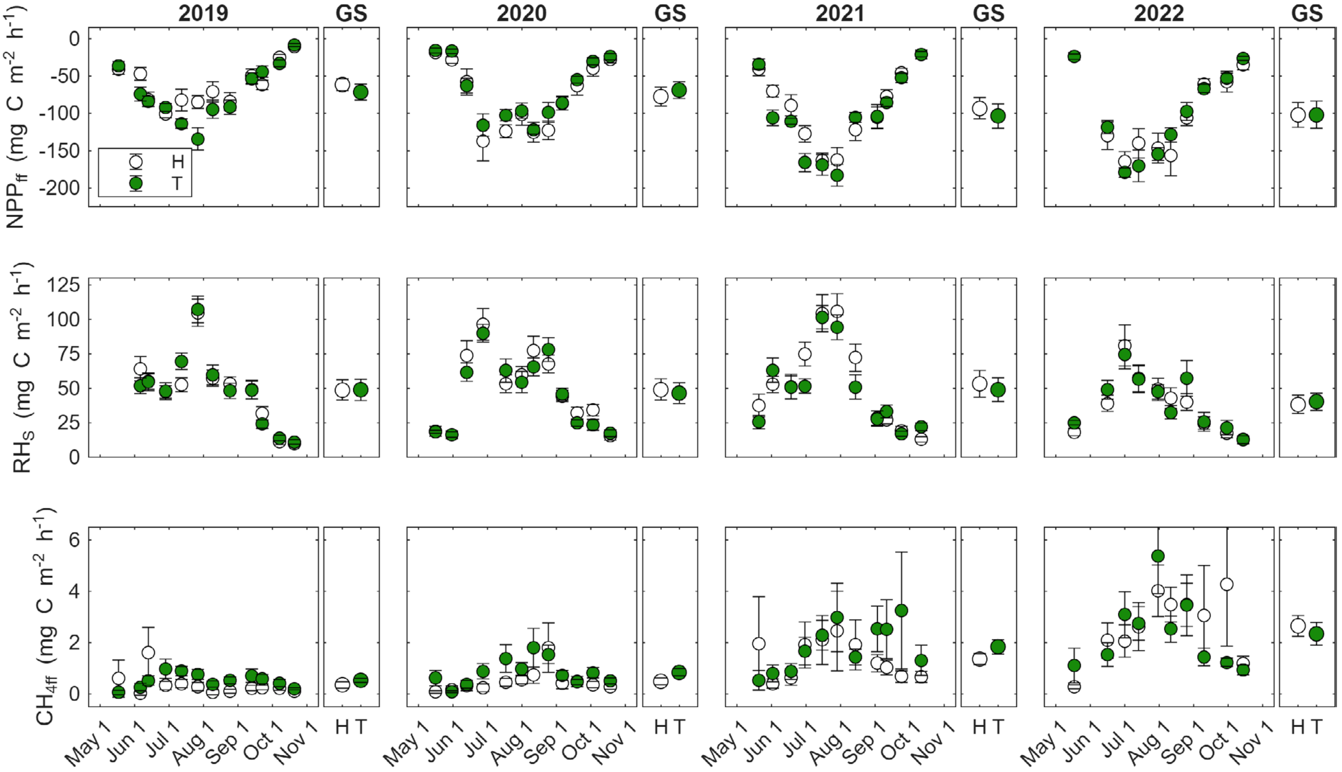

FIGURE 5 | Bi-weekly fluxes during measurement campaigns, including growing season (GS) means of forest floor net primary production (NPP_{ff}, top row), soil heterotrophic respiration (RH_s, middle row), and methane emission (CH_{4ff}, bottom row) for the Harvested (H) and Treed (T) treatments over the drained (2019–2020) and rewetted (2021–2022) years. Error bars are showing one standard error. Note that the H treatment was treed during 2019–2020 and harvested in 2021–2022. A carbon use efficiency of 0.50 was assumed to derive NPP_{ff} as a constant fraction of GPP_{ff}. GPP_{ff} was derived as the balance between measured forest floor net CO_2 exchange (NE_{ff}) and respiration (R_{ff}). Values for the mean measured fluxes are presented in the Table S4.

$m^{-2} year^{-1}$) (Table 1). In the two rewetted years, RH_s was 252 and 219 g C $m^{-2} year^{-1}$ at the H treatment and 242 and 228 g C $m^{-2} year^{-1}$ at the T treatment. Following rewetting and harvest, respiration from decaying tree roots and stumps (RH_r) and branch debris (RH_b) contributed an additional 20 g C $m^{-2} year^{-1}$ to total heterotrophic respiration (RH) in the H treatment.

Annual net CO_2 exchange (NE_p) estimates suggest that the peatland areas in both H and T treatments were net CO_2 sources in 2019, with NEE values of 29 and 25 g C $m^{-2} year^{-1}$, respectively. In 2020, both treatment sites acted as small net CO_2 sinks (–12 and −5.8 g C $m^{-2} year^{-1}$). During the rewetted years, the H treatment site functioned as a net CO_2 source (87 and 53 g C $m^{-2} year^{-1}$ in 2021 and 2022, respectively), while the T treatment site remained a net CO_2 sink (–19 to −55 g C $m^{-2} year^{-1}$).

Annual CH_4 emissions from the forest floor were similar between treatments during the drained years (H: 5.1 and 4.9 g C $m^{-2} year^{-1}$; T: 4.8 and 4.7 g C $m^{-2} year^{-1}$). In the rewetted years, annual CH_4 emissions approximately doubled in both treatments relative to the drained years.

3.4.2 | Ditches

Annual ditch net CO_2 uptake (NE_d) in the drained year 2020 was by an order of magnitude higher than that of the forest floor, with NE_d values of −491 and −465 g C $m^{-2} year^{-1}$ at the H and T transects, respectively (Table 1). Following rewetting, the ditch in the H treatment became a net CO_2 source (140 g C $m^{-2} year^{-1}$) in the first year but returned to being a net CO_2 sink (–174 g C $m^{-2} year^{-1}$) in the second year. In the first rewetting year, annual GPP_d and R_d in the H treatment decreased to 14% and 40%, respectively, of their 2020 values, but recovered to 56% and 68% in the second rewetting year.

Annual CH_4 emissions from the ditch (CH_{4d}) were 22 and 18 g C $m^{-2} year^{-1}$ at the H and T transects, respectively, in 2020, approximately 4 times higher than emissions from the forest floor (Table 1). After rewetting, CH_{4d} emissions in the H treatment decreased to 16 g C $m^{-2} year^{-1}$ during the first year, followed by a 4-fold increase (relative to the drained year) to 79 g C $m^{-2} year^{-1}$ in the second rewetting year.

3.4.3 | Area-Weighted C and GHG Balances

When area-weighting the contributions from the peatland area and ditches, the ecosystem was approximately C neutral averaged over the drained years 2019–2020, with net C balances of −3.3 and −2.0 g C $m^{-2} year^{-1}$ in the H and T treatments, respectively (Figure 7a). The area-weighted contributions of peatland area and ditches to the net ecosystem CO_2 exchange (NEE) were nearly equal, while the area-weighted contribution from

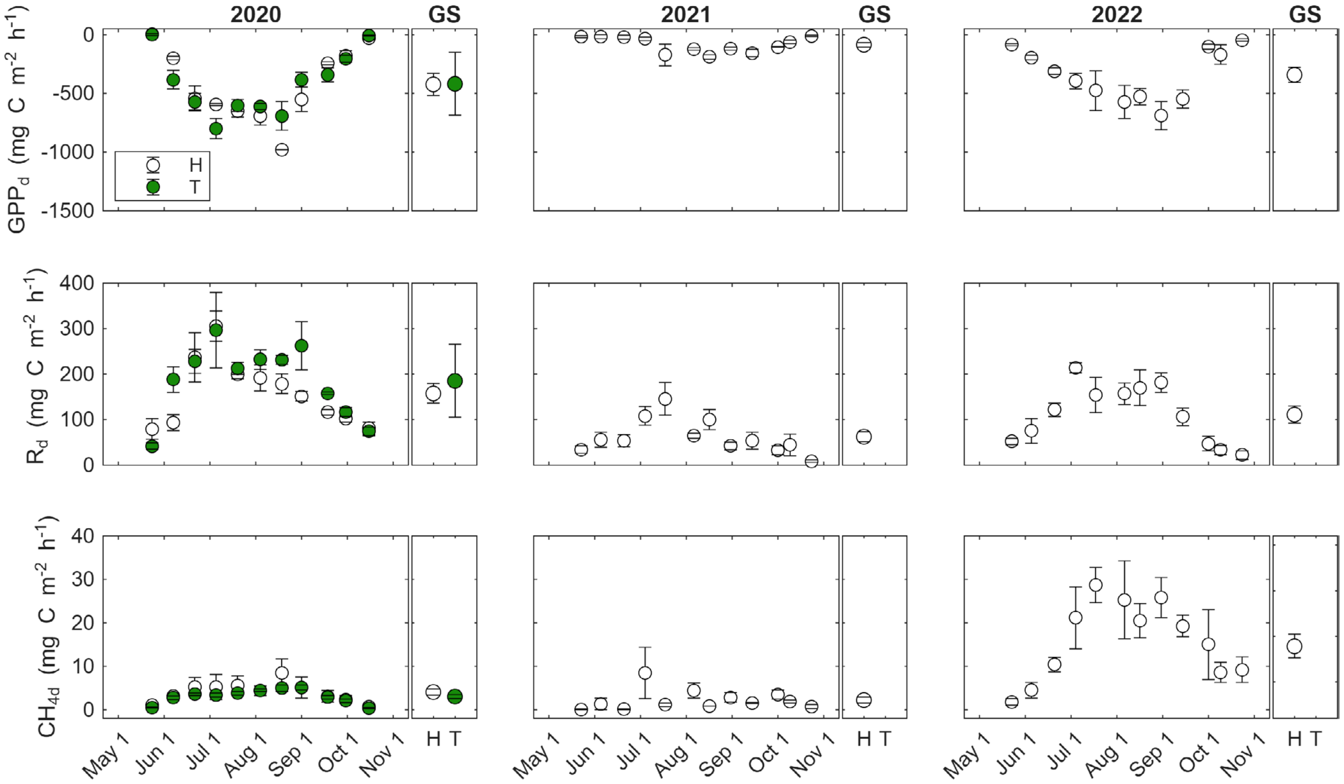

FIGURE 6 | Bi-weekly fluxes during measurement campaigns, including growing season (GS) means of ditch gross primary production (GPP$_d$, top row), respiration (R$_d$, middle row), and methane emissions (CH$_{4d}$, bottom row) for ditch sections in the Harvested (H) and Treed (T) treatments over the drained (2020) and rewetted (2021–2022) years. Error bars are showing one standard error. Note that the ditch section in the H area was drained during 2020 and in-filled during 2021–2022. Data for the unfilled ditch section in the T treatment area during 2021–2022 were not considered. Values for the mean measured fluxes are presented in the Table S4.

the peatland area to ecosystem CH_4 emissions was substantially higher compared to that from ditches.

Following rewetting, the area-weighted sum of peatland area and ditch C balances suggests that the H treatment became a C source of 79 g C m^{-2}year^{-1}, whereas the T treatment was a C sink of −24 g C m^{-2}year^{-1}, averaged over 2021–2022 (Figure 7a). This divergence in the total C balance was primarily driven by contrasting changes in NPP$_{ff}$ and RH between treatments in the peatland area. Furthermore, the contribution of ditches to the ecosystem C balance remained small (1%–5%) during the rewetted years.

When accounting for the GWPs of CO_2 and CH_4, the area-weighted estimates suggest that the drained peatland forest (both H and T treatments) was a net GHG source of 5.3–5.7 t CO_2 eq. ha^{-1}year^{-1} and of 1.6–1.7 t CO_2 eq. ha^{-1}year^{-1} over 20 and 100-year timeframes, respectively (Figure 7b). Following rewetting, GHG emissions increased to 15.0 and 12.4 t CO_2 eq. ha^{-1}year^{-1} in the H and T treatment, respectively, over the 20-year timeframe. Over the 100-year timeframe, the rewetted H treatment was a two-fold larger GHG source (6.7 t CO_2 eq. ha^{-1}year^{-1}) compared to the T treatment (3.3 t CO_2 eq. ha^{-1}year^{-1}). The increase in the GHG emissions following rewetting was largely driven by the increase in CH_4 emissions across both timeframes, while the divergent NEE responses between the H and T treatments contributed to the contrasting GHG balances during the rewetted years.

4 | Discussion

4.1 | Ecosystem C Balance of a Low-Productive Boreal Peatland Forest Before and After Rewetting

Our study provides a detailed in situ assessment of the initial effects of rewetting on the ecosystem C balance and its components, based on pre- and post-rewetting observations in a low-productive, minerogenic boreal peatland forest. Our main results suggest that the ecosystem was, on average, C neutral under pre-rewetting drained conditions. In contrast, during the first 2 years after rewetting, the ecosystems acted as a net C source in the harvested treatment and a net C sink in the non-harvested (treed) treatment. Thus, our study offers valuable empirical evidence to support more accurate evaluations of the potential climate mitigation effect of alternative peatland rewetting strategies.

While our low-productive peatland forest was C neutral in the drained years, findings from a nearby nutrient-poor peatland forest with about 10 times higher standing tree biomass exhibited a considerable net CO_2 uptake (135 g C m^{-2}year^{-1}) during 2020–2022 (Tong et al. 2024). Similarly, other nutrient-poor peatland forests in Fennoscandia (e.g., in Finland) have been reported to act as CO_2 sinks at both the ecosystem and soil level (Lohila et al. 2011; Ojanen et al. 2013; Minkkinen et al. 2018). At our site, however, soil C losses via RH$_s$ likely exceeded the sum of soil C

TABLE 1 | Annual sums (g C m^{-2} $year^{-1}$) of net primary production (NPP) of tree (NPP_t[a]) and forest floor vegetation (NPP_{ff}) components, heterotrophic respiration (RH) from soil (RH_s), branch debris (RH_b), and decaying roots + stumps (RH_r), net CO_2 exchange (NE_p), forest-floor methane emissions ($CH_{4\text{-}ff}$) in the peatland area. For the ditch, gross primary production (GPP_d), respiration (R_d), net CO_2 exchange (NE_d), and methane emissions ($CH_{4\text{-}d}$) are shown. For both surfaces, values are presented for Harvested and Treed treatments during the drained (2019–2020) and rewetted (2021–2022) years.

		Drained				Rewetted			
		2019		2020		2021		2022	
	Flux	Harvested	Treed	Harvested	Treed	Harvested	Treed	Harvested	Treed
Peatland area	NPP	−238.5	−255.4	−253.6	−249.8	−185	−260.9	−184	−282.6
	NPP_t	−46.5	−50.4	−53.6	−49.8	0	−55.9	0	−61.6
	NPP_{ff}	−192	−205	−200	−200	−185	−205	−184	−221
	RH	267	280	242	244	272	242	238	228
	RH_s	267	280	242	244	252	242	219	228
	RH_b	0	0	0	0	9.6	0	9.2	0
	RH_r	0	0	0	0	10.0	0	9.6	0
	NE_p	28.5	24.6	−11.6	−5.8	86.5	−18.9	53.4	−54.6
	CH_{4ff}	5.1	4.8	4.9	4.7	10.2	11.3	10.4	11.6
Ditch	GPP_d	NA	NA	−1272	−1365	−173	NA	−709	NA
	R_d	NA	NA	781	900	313	NA	534	NA
	NE_d	NA	NA	−491	−465	140	NA	−174	NA
	CH_{4d}	NA	NA	22.0	17.7	15.5	NA	79.3	NA

Abbreviation: NA, not applicable.

[a] NPP_t is the sum of changes in tree biomass (aboveground + coarse roots), litterfall and fine root production. Values for these NPP_t components are shown in the Table S6.

inputs from tree litter, ground vegetation litter (even when assuming up to 100% of NPP_{ff} as litter input), and root litter (assumed to equal tree fine root production) in both drained and rewetted years, indicating a net C loss from the peat layer. These results suggest that in peatland forests that are at the lowest end of the productivity range the limited C input may be insufficient to sustain a soil C sink or preserve the peat layer over the long term. In addition, while other drained peatland forests with lower WTL (< -30 cm) have been reported to act as CH_4 sinks (Korkiakoski et al. 2017; Minkkinen et al. 2023; Ojanen et al. 2010), we observed annual CH_4 emissions even during the drained years. This was likely due to the relatively high mean WTL (−26 cm), which further contributed to increase the net GHG emission from the soil.

Our chamber-based estimates of annual NEE at the H treatment during the first two rewetting years (87 and 53 g C m^{-2} $year^{-1}$ in 2021 and 2022, respectively) aligned well with estimates from parallel eddy covariance (EC) measurements (103 and 46 g C m^{-2} $year^{-1}$, respectively) (Tong et al. 2025) carried out during the same years over an area similar to, and partly overlapping with, the H treatment area. In contrast, our chamber-based CH_4 emission estimates (~10 g C m^{-2} $year^{-1}$) for the rewetted H treatment were 2 to 3-fold higher than the EC-based estimates (3.1 and 5.2 g C m^{-2} $year^{-1}$ in 2021 and 2022, respectively) from the same EC-based study (Tong et al. 2025). This discrepancy may be explained by greater vegetation production and the associated substrate supply to methanogens in the eastern section of the rewetted mire, where our study was carried out, compared to the EC footprint, which included larger areas of bare peat with lower CH_4 production potential.

Overall, our results reinforce that a rewetted and harvested boreal peatland forest may not provide immediate climate benefits, as implied by default EFs, which are largely derived from natural mire data in the boreal region (see, e.g., Wilson et al. 2016). Despite contrasting vegetation cover and climate, our observations align with previous studies reporting net C losses from rewetted peat extraction sites in both boreal (Nugent et al. 2018; Strack and Zuback 2013; Tuittila et al. 1999; Yli-Petäys et al. 2007) and temperate (Beyer and Höper 2015; Järveoja et al. 2016) regions within the first decade after rewetting. While our low-productive, nutrient-poor peatland forest site represents a typical feature of the boreal landscape in Sweden, we caution that our results may not be directly transferrable to more productive peatland forests with efficient drainage networks, such as those found in Finland and other parts of Fennoscandia. Additionally, our study is limited in temporal extent, capturing responses only during the initial years following rewetting. Thus, spatially extensive and continued long-term monitoring will be critical to better understand the trajectory of the C sink function and the climate impacts of rewetting boreal peatland forests (Roulet et al. 2025).

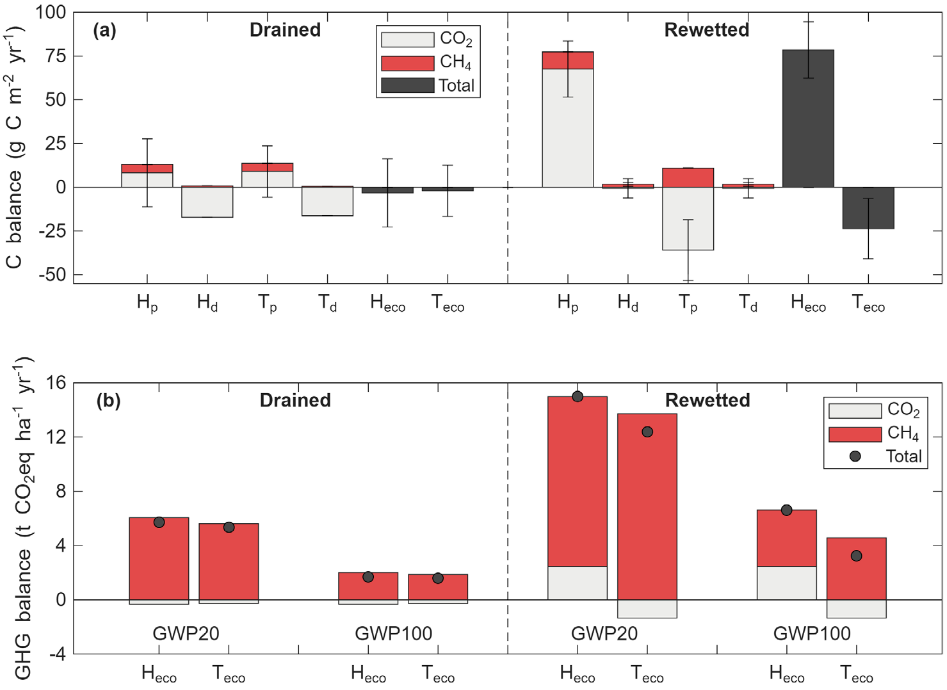

FIGURE 7 | Area-weighted ecosystem-scale (eco) estimates of (a) carbon (C) and (b) greenhouse gas (GHG) balances, integrating fluxes from the peatland area (p) and ditches (d) in the Harvested (H) and Treed (T) treatments averaged over the drained (2019–2020) and rewetted (2021–2022) years. For the rewetted years, the net CO_2 exchange (NEE) and methane (CH_4) fluxes in the T treatment, where the ditch was not infilled, were assumed to be equal to those in the H treatment. GHG balances were calculated by converting fluxes to CO_2 equivalents (CO_2-eq), using global warming potentials of 81 and 27 for CH_4 over 20 and 100-year horizons, respectively. Errors bars in panel (a) denote the standard error of the 2-year averages. Values for the area-weighted fluxes are presented in the Table S5.

4.2 | Initial Rewetting Effects on Annual C Balance Components

Our findings revealed that the peatland forest floor vegetation was the dominant contributor to annual ecosystem C input via NPP in this low-productive, minerogenic peatland forest. Notably, annual production rates of the forest floor vegetation were similar under both drained and rewetted conditions, suggesting no significant short-term impact of rewetting on its production. These results are consistent with a recent study conducted in a nearby (~30 km) drained peatland forest, where forest-floor vegetation productivity was also reported to be a major factor regulating differences in annual ecosystem GPP and NEE between densely and sparsely treed peatland forest stands (Tong et al. 2024). The important contribution of the peatland forest floor vegetation, in particular *Sphagnum* species, in shaping the ecosystem C balance of boreal peatland forests has also been highlighted at other sites across Fennoscandia (Kasimir et al. 2021; Kulmala et al. 2019). Together, these findings underscore the need to account for forest-floor vegetation dynamics when assessing the C balance of drained and rewetted peatland forests. This is essential for producing accurate estimates of the magnitude and direction of the ecosystem C sink function, as well as its sensitivity to climatic and management-induced changes.

Our results also provide new insights into the tree growth response to rewetting. Contrary to the assumption that excess soil water and associated oxygen limitation following rewetting may lead to growth stagnation or tree mortality, we noted a moderate (~18%) increase in tree production in the non-harvested treatment area during the rewetted years. This response may be partially explained by the rewetting-induced moderation of seasonal WTL minima (up to −70 cm under drained conditions), which could have alleviated drought stress and supported increased tree growth during the driest summer months. Alternatively, the observed increase in tree production following rewetting might also be due to reduced growth during the unusually warm and dry conditions of the drained years 2019 and 2020. This interpretation is supported by historical tree ring records, which suggest similar growth rates in the years prior to 2019 relative to the rewetting years (Figure S2). Nevertheless, a better understanding of long-term tree growth responses to peatland rewetting is needed to improve predictions of C balance trajectories and associated climate impacts of rewetting peatland forests without tree harvest.

Our results further suggested that annual soil RH decreased following rewetting, although the effect size (~13%) was limited and only evident in the second rewetting year. The limited response was likely due to the relatively high mean WTL (−26 cm) already present at our site during the drained conditions. Thus, this finding supports previous studies suggesting that the potential for soil RH mitigation and associated climate benefits from rewetting peatland forests that are poorly drained (due to a gradual decrease of the ditch drainage function over time) might be small, relative to well-drained (and nutrient-rich) sites in other regions (Ojanen and Minkkinen 2020; Laine et al. 2024; Launiainen et al. 2025). In consequence, while low-productive, forested peatland areas are a dominant feature of the

Swedish boreal landscape, they may not be optimal target areas for achieving substantial soil emission reductions via rewetting measures in the coming decades.

The observed increase in forest floor CH_4 emissions during the rewetted years aligns with the well-established understanding that higher WTL and associated anoxic conditions lead to a significant increase in CH_4 emissions following rewetting (Cooper et al. 2014; Koskinen et al. 2016; Vanselow-Algan et al. 2015; Wilson et al. 2009). When accounting for gas-specific GWPs, our results further highlight that CH_4 emissions were the primary driver of the initial warming climate impact following rewetting. Thus, from the perspective of evaluating the full climate impact of rewetting, and its potential to contribute to the climate goal of net zero emissions by 2045, it is critical to fully understand the consequences of rewetting on CH_4 emissions and its counterbalancing effects on the benefit of reduced soil CO_2 emissions.

Our results further demonstrate the considerable contribution of ditch CO_2 and CH_4 fluxes to the ecosystem-scale C and GHG balances. Under drained conditions, the area-weighted net CO_2 uptake by extensive vegetation growing in the unmaintained ditches exceeded the net CO_2 loss from the peatland area, resulting in an ecosystem-scale NEE that was close to C neutral. It is noteworthy that considerable CO_2 uptake might be a unique feature of poorly maintained and subsequently in-grown ditches, compared to well-maintained drainage networks which commonly act as CO_2 sources (Hyvönen et al. 2013). The ~15% contribution of ditch CH_4 fluxes to total ecosystem CH_4 emission further highlights the need to account for ditch fluxes when estimating total C and GHG balances of drained peatland forests. These findings are in line with previous studies that identified drainage ditches in peatland ecosystems as potential hotspots for CO_2 and CH_4 fluxes (Cooper et al. 2014; Minkkinen et al. 1997; Peacock, Audet, et al. 2021; Peacock, Granath, et al. 2021; Waddington and Day 2007; Xue et al. 2025). Considering this, there is a further need to also understand how ditch CO_2 and CH_4 fluxes respond to infilling during rewetting actions. Our study demonstrates that vegetation removal and soil disturbance during infilling transformed the infilled ditch into a significant CO_2 source in the first year after rewetting. However, by the second rewetting year, the infilled ditch returned to a net CO_2 sink, with uptake reaching about 35% of the pre-filling levels. This suggests that the disturbance effect on ditch NEE might be transient, likely due to relatively rapid establishment and/or recovery of forest-floor vegetation within the infilled ditch area.

In contrast to the ditch NEE response, the manifold increase in CH_4 emission by the second year after rewetting raises concerns that infilled ditches may become major hotspots for CH_4 emissions in rewetted peatlands. At our site, this pronounced rise in ditch CH_4 emissions was likely caused by the rapid establishment of *E. vaginatum*, a species known to facilitate transport for CH_4 from anaerobic peat layer into the atmosphere (Greenup et al. 2000). Nevertheless, long-term monitoring and additional observations from other boreal rewetting sites are essential to better understand the potential role of CH_4 emissions from infilled ditches in shaping the trajectories in the ecosystem-scale C and GHG balances of rewetted peatland forests.

4.3 | Alternative Tree Harvest Management Regulates the Climate Impact of Rewetting

To our knowledge, this study provides the first empirical in situ evidence that tree-harvest decisions can significantly shape the initial climate impact of peatland rewetting. Our results suggest that preserving the existing tree layer during rewetting offers immediate benefits for the ecosystem-scale C sink and GHG balance. Specifically, the continued CO_2 uptake by the remaining trees, combined with the avoided respiration from harvest residue, considerably reduced GHG emissions (relative to the harvested area) during the initial rewetted years. However, while our study focused on short-term responses, continued monitoring is necessary to understand long-term changes in tree growth rates under rewetted conditions. Furthermore, the greater C and GHG emissions observed in the harvested treatment were partially attributable to increased C losses via RH from decaying tree root systems and slash debris left on site after harvest. It is noteworthy that our approach for quantifying soil RH from trenched (i.e., root-free) plots could not capture effects of reduced root-derived C supply to heterotrophic decomposer communities following harvest. Despite this shortcoming, our study highlights the influence of tree harvest on the ecosystem CO_2 balance. Given that our study site was a low-productive peatland forest, it is likely that such harvest-related effects may be even more pronounced in more productive peatland forests undergoing rewetting.

In contrast to the effects on the CO_2 balance, tree harvest did not significantly modulate the CH_4 emissions response to rewetting at our low-productive site. However, in more productive peatland forests, retaining the tree layer may sustain higher transpiration rates following rewetting, potentially lowering the WTL and thereby reducing the CH_4 emission rates compared to rewetted areas where the trees have been harvested (Laine et al. 2024). Considering the 81 times higher GWP of CH_4 relative to CO_2 over a 20-year timeframe, a detailed understanding of harvest-related changes in CH_4 emissions is critical for accurately assessing the climate impact from rewetted peatland forests.

While the accounting boundary in this study was limited to the ecosystem scale, it is important to recognize that consequences of harvest decisions for the climate impact of rewetting also encompass the effects of emission reduction from substituting fossil fuel-based material and long-term C storage in wood products derived from harvested biomass (Launiainen et al. 2025; Ojanen and Minkkinen 2020). These substitution and storage effects are likely to become increasingly important in the context of rewetting areas with high-productive peatland forests. Furthermore, tree harvest can influence local and regional climate through biophysical effects, including changes in surface roughness, albedo (i.e., reflectance of solar radiation energy), partitioning of sensible and latent heat fluxes, and emission of secondary organic compounds (SOCs; as precursors for aerosols) (Kalliokoski et al. 2020; Launiainen et al. 2025). Thus, holistic approaches will be required to develop a comprehensive understanding of the net climate impacts of alternative peatland rewetting strategies.

Author Contributions

Järvi Järveoja: conceptualization, methodology, formal analysis, investigation, resources, visualization, data curation, supervision, funding acquisition, project administration, writing – original draft. **Alexander Pinkwart:** formal analysis, investigation, data curation, visualization, writing – review and editing. **Cheuk Hei Marcus Tong:** formal analysis, data curation, writing – review and editing. **Eduardo Martínez-García:** methodology, writing – review and editing, investigation. **Hjalmar Laudon:** resources, writing – review and editing. **Matthias Peichl:** conceptualization, writing – review and editing, methodology. **Mats B. Nilsson:** conceptualization, writing – review and editing.

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Conflicts of Interest

The authors declare no conflicts of interest.

Data Availability Statement

The data supporting the findings of this study are openly available in the Zenodo digital repository at https://doi.org/10.5281/zenodo.20385091. This study does not include any code.

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Supporting Information

Additional supporting information can be found online in the Supporting Information section. **Figure S1:** Box plots of instantaneous PAR measurements above and below the forest canopy during the rewetted years (2021–2022) in the harvested (H) and treed (T) treatments. **Figure S2:** Annual tree ring increments at the Treed treatment during 2013–2022. The vertical dashed line denotes the start of the study period, with the shaded and white panel areas showing the drained (2013–2020) and rewetted (2021–2022) years, respectively. Light gray bars represent the interquartile ranges (IQRs), the black line indicates the median, and colored points represent mean increments per plot ($n = 12$, with 2 plots being treeless). Note that two plots have exceptionally high tree growth due to shallower peat depths. **Table S1:** Model parameters used for upscaling of peatland forest floor fluxes during the growing season, including gross primary production (GPP$_{\text{ff}}$), soil heterotrophic respiration (RH$_{\text{s}}$) and methane fluxes (CH$_{4\text{ff}}$) in drained (2019–2020) and rewetted (2021–2022) years at the Harvested and Treed treatments. Significance levels of parameters are indicated with asterisks: *$p < 0.05$, **$p < 0.01$, ***$p < 0.001$. R^2 and RMSE represent the model coefficient of determination and the root mean square error, respectively. Model equations and parameters are described in the subsection 2.6.1 in the main manuscript. **Table S2:** Model parameters used for upscaling of soil heterotrophic respiration (RH$_{\text{s}}$) during the non-growing season in drained (2019–2020) and rewetted (2021–2022) years at the Harvested and Treed treatments. Significance levels of parameters are indicated with asterisks: *$p < 0.05$, **$p < 0.01$, ***$p < 0.001$. R^2 and RMSE represent the model coefficient of determination and the root mean square error, respectively. Model equations and parameters are described in the subsection 2.6.1 in the main manuscript. **Table S3:** Model parameters used for upscaling of peatland ditch fluxes to annual estimates, including gross primary production (GPP$_{\text{d}}$) and ecosystem respiration (ER$_{\text{d}}$) in drained (2020) and rewetted (2021–2022) years at the Harvested and Treed treatments. Significance levels of parameters are indicated with asterisks: *$p < 0.05$, **$p < 0.01$, ***$p < 0.001$, ns = not significant. R^2 and RMSE represent the model coefficient of determination and the root mean square error, respectively. Model equations and parameters are described in the subsection 2.6.2 in the main manuscript. **Table S4:** Growing season means of the measured fluxes presented in Figures 5 and 6 in the main manuscript. **Table S5:** Values for the area-weighted fluxes presented in Figure 7 in the main manuscript. **Table S6:** Annual values of the tree net primary production (NPP$_{\text{t}}$) components, including changes in tree biomass (aboveground and coarse root), litterfall and tree fine root production (FRP) in the drained and rewetted years for Harvested and Treed treatments.