









Carbon dynamics after thinning in two boreal forest sites: Upland and drained peatland

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ARTICLE INFO

Keywords:

Forest management
Thinning
Carbon budget
Net ecosystem production
Carbon Balance
Boreal forest

ABSTRACT

Boreal forests act as crucial carbon storage, but their management can have important consequences for their carbon dynamics. Thinning of the stand alters carbon storage by removing trees, which affects the overall forest growth. Thinning stimulates growth of the remaining trees and affects soil processes in complex ways, while the overall impacts vary by forest type and site conditions. We applied biometric and soil CO₂ flux measurement data to study the changes in carbon storage across multiple forest components in both an upland and a drained peatland forest undergoing thinning operations. Following thinning, the carbon accumulation rate of the aboveground biomass components declined sharply due to the removal of trees but started recovering a year after the thinning, largely driven by forest floor vegetation. Belowground biomass components followed a similar trend. Carbon emissions increased post-thinning, mainly due to decomposition of harvest residues. The upland forest net ecosystem production (NEP) temporarily shifted to a net source of carbon (-43 g C m⁻² yr⁻¹) but it recovered the following year to a net carbon sink (164 g C m⁻² yr⁻¹). The drained peatland forest NEP remained negative, with thinning further increasing the CO₂ emissions (-570 g C m⁻² yr⁻¹) and leading to a slow recovery (-488 g C m⁻² yr⁻¹) to pre-thinning levels. The tree carbon stocks in both forest type is projected to take over a decade to recover. The effects of thinning on forest carbon dynamics show short-term changes in the upland forest but more lasting consequences in the drained peatland forest due to slower tree growth and persistent emissions.

1. Introduction

The boreal forest, spanning vast regions across the northern hemisphere, is a crucial component of the global carbon cycle. Characterized by cold climates, short growing seasons, and diverse soil conditions, boreal forests serve as a substantial carbon sink, storing large quantities of carbon in both plant biomass and soil (Deluca and Boisvenue, 2012; Liski et al., 2006; Milakovskiy et al., 2012). Their ability to sequester carbon plays a vital role in mitigating climate change. The carbon sequestration of boreal forests is greatly affected by forest management

that shapes carbon cycling in these ecosystems. Nearly two-thirds of boreal forests are subject to some form of management, particularly for wood production (Gauthier et al., 2015), with this proportion being even higher in the Nordic countries.

Thinning is a common silvicultural practice in boreal Fennoscandia (Kuuluvainen et al., 2012) with the aim of allocating growth to the fewer, remaining trees and enhance timber yield for sawn wood production. In Finland, approximately 70 % of forest land is classified as upland (mineral soil) forest, while around 30 % consists of peatland forests, many of which have been drained for forestry purposes (Natural

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<https://doi.org/10.1016/j.foreco.2025.123024>

Received 12 June 2025; Received in revised form 16 July 2025; Accepted 17 July 2025

Available online 4 August 2025

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Resources Institute Finland (Luke), 2020). The influence of thinning on the carbon budget of a forest ecosystem is complex, as it reshapes carbon dynamics across multiple ecosystem components, including trees at various canopy layers, forest floor vegetation, and soil. Thinning directly reduces aboveground biomass by removing tree stems, but it also stimulates the growth of the remaining trees and understory vegetation, offsetting initial carbon losses over time. The net effect of thinning on forest carbon dynamics varies depending on forest type, climate, and soil conditions (Mäkipää et al., 1999; Vesterdal et al., 1995). In a short-term, thinning typically reduces annual forest carbon accumulation by removing trees whose foliage area contribute to carbon sequestration (Aun et al., 2021a, 2021b), thereby decreasing the forest's sink capacity. Although thinning initially reduces forest carbon uptake by removing foliage and biomass, several studies report that growth of residual trees often increases in subsequent years, promoting a gradual recovery of carbon accumulation (Hoover and Stout, 2007; Mäkipää et al., 1999; Meng et al., 2022). This recovery is attributed to reduced competition for light and nutrients, which improves growing conditions and allows compensatory growth in both the understory and remaining canopy trees. In upland boreal forests, the carbon sink strength may return to pre-thinning levels within a few years (Lindroth et al., 2018; Aslan et al., 2024). However, in drained peatland forests, research on the effects of thinning on the carbon budget remains limited, and responses to thinning may differ from that happening in mineral soils due to high soil organic matter content and partly waterlogged conditions (Straková et al., 2012; Munir et al., 2014). Additionally, thinning may reduce plant competition and increase light and moisture availability, promoting understory growth (Thomas et al., 1999). These changes can also alter forest floor vegetation composition (Lindh and Muir, 2004; Ares et al., 2009), which may, in turn, influence carbon fluxes in both aboveground and belowground compartments.

Thinning also affects soil respiration rates, with the magnitude of this effect varying based on site-specific conditions (Vesala et al., 2005; Aslan et al., 2024; Lindroth et al., 2018). In upland forests, thinning often leads to elevated soil heterotrophic respiration in the short term due to increase in the number of dead roots, stumps, and litter in the soil and increase in the soil temperature from greater sunlight exposure (Lei et al., 2018). In drained peatlands, thinning can raise the water table level (Peltoniemi et al., 2023; Leppä et al., 2020), which reduces soil aeration and can decrease decomposition rates (Ise et al., 2008; Laiho, 2006). However, a functioning ditching system can regulate the water table levels, leading to an increase of the decomposition of dead root biomass and organic matter. This combined with soil warming could lead to increased soil respiration. Soil carbon fluxes, both carbon dioxide and methane, change depending on site conditions with methane emissions potentially increasing due to thinning in sites with high water table level such as peatlands (Korkiakoski et al., 2020). Another consequence of thinning in both forest types is the high amount of harvest residues, such as foliage and branches left in the forest. This contributes to increased carbon emissions through decomposition (Palviainen et al., 2004; Vesala et al., 2005). These processes trigger cascading effects on the ecosystem's carbon balance, underscoring the complexity of quantifying thinning's overall impact on carbon dynamics.

Numerous studies (Zhang et al., 2018; Garcia-Gonzalo et al., 2007) have examined the effects of thinning on different components of a forest stand, but these components are rarely analyzed together within the same temporal and spatial scale, particularly in drained peatland forests. Over the past two decades, research on forest carbon budget has increasingly relied on the eddy covariance (EC) method. This technique provides continuous, ecosystem-scale estimates of the net ecosystem exchange of a greenhouse gas under investigation (Baldocchi, 2003; Eugster and Merbold, 2015). However, EC has limitations in analyzing individual forest components, as it provides only an integrated signal of net fluxes and covers typically a large area of several hectares. For instance, while it can estimate total forest gross primary productivity, it

cannot distinguish whether carbon is being accumulated more in foliage or in stems, or in trees or forest floor vegetation. To address this gap, a biometric approach combined with soil carbon dioxide (CO₂) exchange measurements can be employed to comprehensively evaluate carbon accumulation responses across multiple forest components. Biometric measurements are regarded as a reliable method for estimating net primary productivity (Curtis et al., 2002; Luysaert et al., 2009). When combined with forest floor gas exchange measurements and upscaled to the stand level, they can provide a forest carbon budget comparable to EC-based estimates.

In this study, we estimate how thinning influences carbon dynamics across multiple forest components, including aboveground and belowground tree biomass, forest floor vegetation, soil respiration, and deadwood decomposition, using two study sites representing contrasting forest types: one on an upland forest and one on a drained peatland forest. The estimates are also compared to previously published EC results from these same or similar sites (Tikkasalo et al., 2024; Aslan et al., 2024; Korkiakoski et al., 2023). We hypothesize (H1) that the carbon accumulation rate of forest components will initially decrease following thinning but recover to pre-thinning levels in few years. We also hypothesize (H2) that post-thinning, carbon emissions will increase due to enhanced biomass decomposition from harvest residues, which will start to decrease a year after thinning. Finally, we hypothesize (H3) that the drained peatland forest will exhibit higher carbon emissions than the upland forest, primarily due to accelerated decomposition of abundant soil organic matter through increased heterotrophic respiration, along with the decomposition of harvest residues, leading to a slower recovery of net ecosystem production to pre-thinning levels.

2. Materials and methods

2.1. Forest site descriptions

The two studied forest sites were located in southern Finland: an upland forest (UP) at the SMEAR II station in Hyytiälä (N 61°50'43'', E 24°17'13''), and a drained peatland forest (DP) at the Ränskälänkorpi experimental site (N 61°10'58'', E 25°15'59''). The two study sites differ in forest type, main tree species, management history, and environmental conditions. To facilitate comparison, the key characteristics of each site are summarized in Table 1 and Fig. 1. More information

Table 1

Summary of forest stand characteristics, management history, and environmental conditions at the two study sites: Hyytiälä upland forest (UP) and Ränskälänkorpi drained peatland forest (DP). Data correspond to the study period. Sources: kolari et al. (2022), Aalto et al. (2023), and laurila et al. (2021).

Characteristic	Hyytiälä (UP)	Ränskälänkorpi (DP)
Forest type	Medium-fertility upland	Nutrient-rich drained peatland
Dominant species	<i>Pinus sylvestris</i> (Scots pine)	<i>Picea abies</i> (Norway spruce)
Other species	<i>Picea abies</i> , <i>Betula pendula</i> , <i>B. pubescens</i>	<i>Pinus sylvestris</i> , <i>Betula pubescens</i>
Stand age / origin	Established in 1960s	Drained pre-1960s
Thinning year	2020	2021
Basal area before thinning (m ² /ha)	31	33
Basal area after thinning (m ² /ha)	18 (40 % reduction)	14 (58 % reduction)
Biomass removed (%)	40 %	75 % of dominant tree biomass
Control area size (ha)	0.6	7.3
Thinned area size (ha)	10	10
Air temperature (mean, °C)	4.2–6.3 (2019–2022)	4.7–6.6 (2020–2022)
Air temp range (°C)	−21.0–26.0	−19.7–26.4
Soil temperature (°C)	6.0–6.6	5.5–6.1
Soil moisture (%)	28–35 %	29–30 %
Precipitation (mm)	399–834	568–671

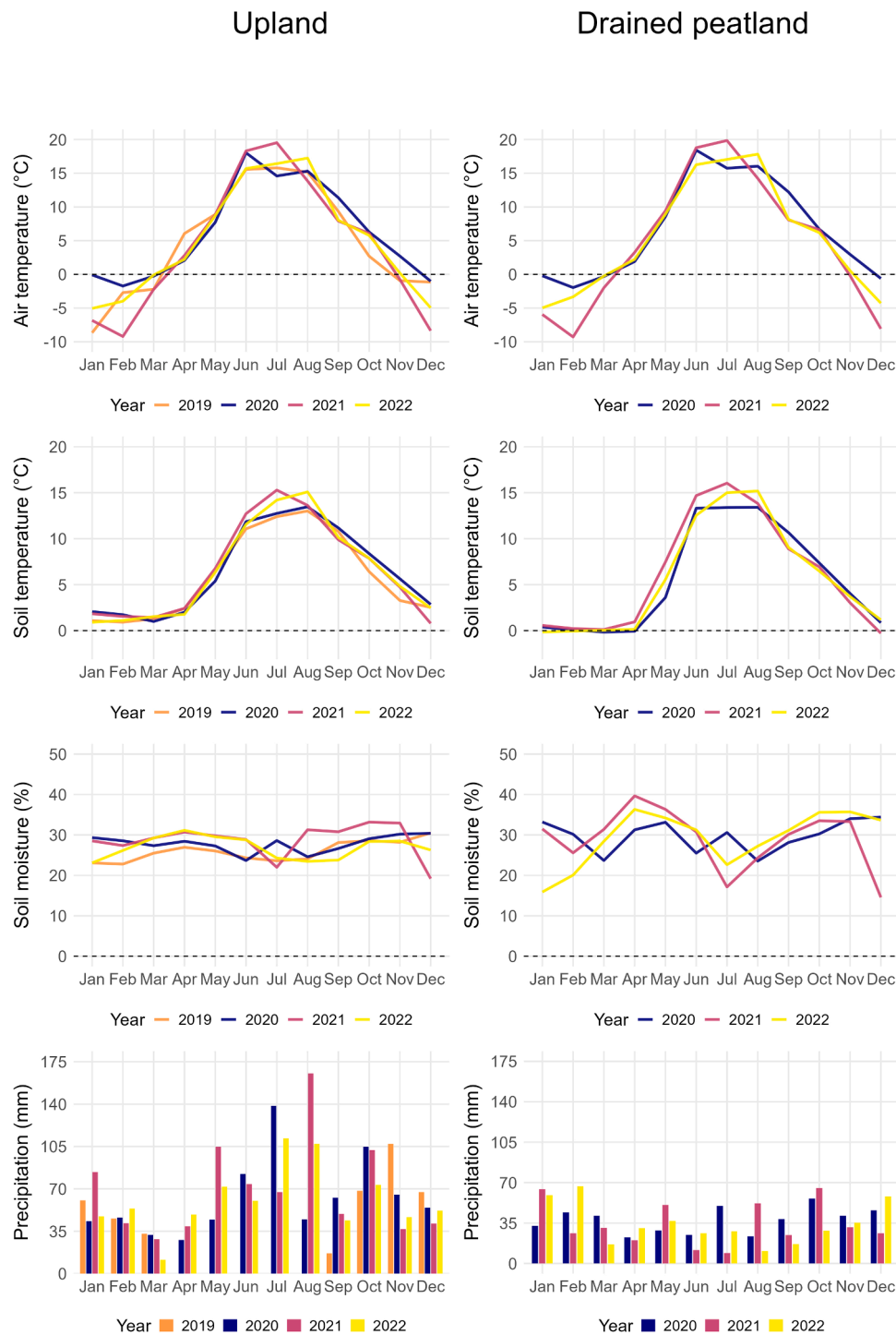


Fig. 1. Meteorological conditions in the upland forest (Hyytiälä) and the drained peatland forest (Ränskälänkorpi) during each year of the study. Panels show monthly air temperature, monthly soil temperature in the A horizon (between 0 and 5 cm depth), monthly soil moisture in the A horizon, and monthly precipitation. The thinning was done in the beginning of the year 2020 in Hyytiälä and 2021 in Ränskälänkorpi.

regarding the study sites can be found in [Aalto et al. \(2023\)](#) and [Laurila et al. \(2021\)](#).

2.2. Carbon budgeting

The carbon budget was determined by balancing the inputs and outputs of carbon for both control and treated sites at each study location across three time periods: pre-thinning, the year of thinning, and post-thinning (shown in [Fig. 2](#)). This assessment utilized the net

ecosystem production (NEP) concept, defined as the difference between net primary production (NPP) and carbon losses, including heterotrophic respiration (Rh), decomposition of deadwood and residues (D), and leaching (L) ([Eq. 1](#)) ([Lovett et al., 2006](#)). In this study, NPP was calculated as the sum of the net growth of all tree components and forest floor vegetation. Leaching was estimated based on values reported by [Räty et al. \(2020\)](#) for UP and by [Rantakari et al. \(2010\)](#) for DP. The biomass removed by thinning was not included in the NEP estimates, allowing for comparison with previous studies that used the eddy covariance

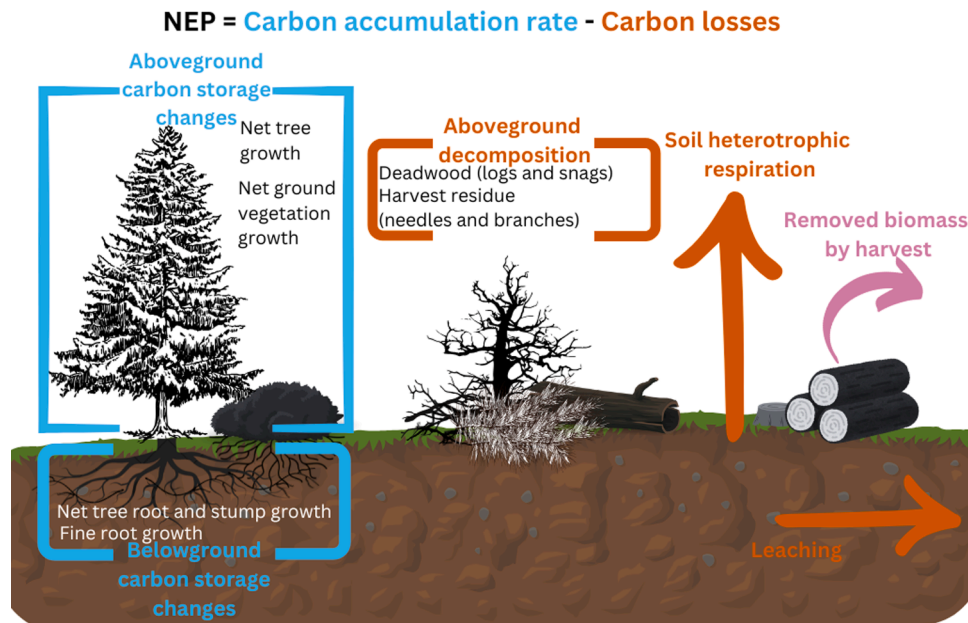


Fig. 2. Schematic of the net ecosystem production (NEP) calculations, where NEP represents the balance between carbon accumulation rate through above and below ground growth and carbon losses through decomposition, soil heterotrophic respiration, and leaching. The biomass removed from the site after thinning represents a carbon loss from the system but is not included in the NEP calculations.

method.

$$NEP = NPP - (Rh + D + L) \quad (1)$$

NEP represents the rate of carbon accumulation in a forest ecosystem and characterizes whether a forest is a carbon sink or source. In this study, we used the ecological definition of NEP, where a positive NEP value indicates that the ecosystem behaves as a carbon sink, while a negative value signifies a carbon source. These results were compared to previous studies on the same locations that used EC to assess the net ecosystem exchange of the forest ecosystem (Aslan et al., 2024) and results in a similar drained peatland forest (Laurila et al., 2021; Lehtonen et al., 2025a).

2.3. Above and belowground tree carbon accumulation rate

2.3.1. Forest site inventories

The UP's thinned site was inventoried before the thinning in 2020 (details in Aalto et al., 2023), while the DP (control and thinned sites) was inventoried between 2019 and 2020 (details in Laurila et al., 2021). These inventories recorded tree species, status (alive or dead), diameter at breast height, total height, crown length, and whether a tree was marked for removal or designated to remain at the thinning site. The unthinned control site in the UP was inventoried later, in 2022, with four plots (each 25 m in diameter) established for measurements. As in previous inventories, all trees within these plots were measured, and their species, size and status were recorded.

2.3.2. Tree aboveground growth

To compare tree biomass growth, tree diameter and height increments were measured. In late summer 2022, increment cores were taken from 80 trees per forest type (40 control, 40 treated site). Following Aakala (2018), the cores were glued and sectioned with a microtome to expose the growth rings, scanned, and analyzed with WinDendro software (Regent Instruments Inc., Québec, Canada). The diameter growth was grouped by tree species and 10 cm diameter classes, with missing values estimated as averages of the adjacent classes. Annual height growth was estimated from trees measured in a sampling plot using a PM5-1520PC hypsometer (Suunto, Vantaa,

Finland), while crown length changes were determined by the ratio of total height to crown height.

2.3.3. Biomass and carbon accumulation rate estimations using allometric equations

Using these data on stem diameter, tree height and crown height of each studied year, biomass was estimated for tree stem, bark, branches, stump, and coarse roots using species-specific allometric equations (Repola, 2008 (model 1a for stem and bark, 1b for stump and coarse roots, and 2 for branches); Repola, 2009 (model 1a,c for stem and bark, 1b,d for stump and coarse roots, and 2a,b for branches)), while foliage biomass was calculated based on pipe model theory (Lehtonen et al., 2020a, 2020b; Tupek et al., 2015). Biomass growth for each year was estimated as the difference in component biomass between consecutive years.

Because needle turnover is continuous, yearly foliage biomass changes underestimate growth. Using monthly litter trap data, annual needle loss was estimated. Based on species needle retention (three years for Scots pine, five years for Norway spruce), we assumed annual needle replacement of one-third in UP and one-fifth in DP. These estimates were included in the total annual foliage biomass growth. Carbon content in the biomass of stems, bark, branches, stumps, and coarse roots was estimated by multiplying the estimated dry biomass by species-specific carbon concentrations reported by Mensah and Petersson (2024) for Svartberget, a site geographically close in latitude to the forests in this study. Foliage carbon content was assumed to be 50 % of dry biomass for all species.

2.3.4. Fine root growth

Fine root growth was measured using ingrowth cores with 1 mm mesh polyester fabric installed in control and treated sites of both forest types. In the UP, 28 cores ($\varnothing 40 \text{ mm} \times 400 \text{ mm}$) filled with sieved mineral soil were installed in early 2020 and collected after two years. In the DP, ten cores ($\varnothing 32 \text{ mm} \times 500 \text{ mm}$) filled with sieved peat substrate were installed in May 2021 and retrieved in November 2022. Roots were manually sorted, dried, and weighed following Laiho et al. (2014). Annual fine root growth was estimated as half of the two-year total, assuming constant growth.

2.3.5. Forest floor vegetation growth

Forest floor vegetation (vascular plants and mosses) was destructively sampled in both forest types at the end of the 2021 and 2022 growing seasons. In the UP, shrubs and herbaceous vegetation was collected from 50 × 50 cm plots, with a 25 × 25 cm subplot at the center for moss sampling. In total, 12 plots per site (6 control, 6 treated) were installed in 2021 and 20 per site (10 control, 10 treated) in 2022. Annual biomass growth was estimated from newly produced leaves and shoots. For example, only newly formed foliage was collected from evergreen *Vaccinium vitis-idaea* L., while total leaf biomass was considered as annual growth for deciduous *Vaccinium myrtillus* L. The biomass was oven-dried and weighed. Similarly, moss biomass was collected from the smaller subplots, with new growth separated under a microscope before drying and weighing.

In the DP, vegetation was collected from 30 × 30 cm plots in late summers of 2020–2022 (7 control, 12 treated). Shrubs and herbaceous vegetation annual growth was separated in the lab and oven dried. Moss was collected from 25 × 25 cm subplots, with 12 plots per site in 2021 and 20 in 2022, following the same method used in UP. The carbon content of fine roots, forest floor vegetation, and moss was assumed to be 50 % of their dry biomass.

In the absence of pre-thinning data for the UP, the growth of fine roots and forest floor vegetation (including shrubs, herbaceous vegetation, and moss) was assumed to be similar between the control and treated areas in the year prior to thinning. The same assumption was applied to fine root and moss growth in the peatland forest.

2.4. Aboveground decomposition of litter and harvest residues

Aboveground decomposition was defined as the decay of deadwood (logs and snags) and harvest residues (leaves and branches). Carbon emission from deadwood decomposition were estimated from the previously mentioned tree inventory data which registered the standing and fallen dead trees in each forest type. In the UP, the control was inventoried separately to register standing and fallen deadwood. We used the biomass models previously introduced, and then converted the biomass to carbon using the methods outlined earlier. Carbon emissions from snags and logs were estimated using modeled stem and bark biomass, applying an exponential decay function (Olson, 1963) to calculate annual carbon loss (Eq. 2). In this equation, Y represents the remaining mass, Y₀ is the initial mass, and t is the time in years. Decomposition rates (k) for snags and logs were derived from Yatskov et al. (2003). This same equation was applied to estimate the decomposition of the removed tree stumps.

$$Y = Y_0 e^{-kt} \quad (2)$$

Carbon emissions from harvest residue were estimated using litter bags (10 × 10 cm, 1 mm mesh) containing 2.5 g of oven dry needles collected in spring 2020 from both forest type. A total of 84 bags were placed in control and treated sites of the UP and DP in early 2020, with 28 bags collected annually per treatment. The bag contents were dried and weighed to determine biomass loss. To estimate carbon emissions from harvest residue, initial branch samples were collected from residue piles to determine their dry weight. Subsequently, 12 branch samples were collected annually in 2021 and 2022 to measure biomass loss. The bulk density of branch samples collected in 2020 in the UP was measured via water displacement and compared with Lintunen et al. (2011) to estimate mass changes for that year. These data were used to calculate annual carbon emissions from harvest residue decomposition by applying mass loss to the carbon content of branches and foliage from the harvested trees.

2.5. Carbon removed in thinning

The amount of carbon removed from the forest stand during thinning

was estimated using data from the forest inventories, which identified the trees marked for removal. This calculation followed the same methodology used to estimate aboveground tree carbon (see 2.2.3), focusing on the carbon content of the stem and bark, which was the biomass that was transported away from the site.

2.6. Soil heterotrophic respiration

2.6.1. Upland forest (UP)

Two setups to estimate the net CO₂ exchange from soil. From 2019–2020, measurements were taken using three transparent automatic flow-through chambers, as described in detail by Aaltonen et al. (2013). However, by the end of 2020, the setup was replaced by six transparent automatic chambers, following the description and measurements in Koskinen et al. (2014), with some modifications that are described in the supplementary material S1. The carbon fluxes inside the chambers were derived from the mass balance equation described by Hari et al. (1999).

Nighttime measurements from these chambers were used to model dark respiration with an adjusted version of the Lloyd and Taylor (1994) equation, which accounts for, in addition to soil temperature, the effects of relative soil water content (RWC) by incorporating a scaling term derived from Ryhti et al. (2021) (Eq. 3). Total forest-floor respiration (R) was parameterized using RWC and soil temperature through a non-linear least squares model (Eq. 3), where base respiration (R₀), alpha, and E are constants determined by the model, and T₀ and T₁ where constants used previously by Korkiakoski et al. (2019).

$$R = \left[1 - \left(\frac{1 - RWC}{\alpha} \right)^v \right]^{-1} R_0 * e^{E \left(\frac{1}{T_0} - \frac{1}{T_{soil} - T_1} \right)} \quad (3)$$

Wintertime respiration was estimated from the same automatic chambers and measurements from six manual opaque chambers conducted at 4-week intervals. The specifications of the manual chambers can be found in Lindfors et al. (2025).

Once forest-floor dark respiration was determined, it was partitioned into its components based on a soil trenching experiment conducted at the same site by Ryhti et al. (2021): 30 % heterotrophic respiration, 48 % tree root respiration, and the remainder attributed to ground vegetation. These proportions were used to estimate heterotrophic respiration in both the control and pre-thinning treated areas of the UP. However, tree removal is expected to alter the balance between autotrophic and heterotrophic respiration (Lei et al., 2018). To account for this, we assumed that 40 % of tree root respiration ceased after thinning, as 60 % of root biomass was estimated to remain. Based on modeled belowground biomass, we assumed a 6 % annual regeneration rate for coarse roots, resulting in a gradual recovery of tree root respiration. Forest floor vegetation respiration was assumed to remain unchanged immediately after thinning but to increase by 75 % the following year, based on observed biomass increases, and to remain at that level in the second year. Since plant respiration is generally proportional to biomass (Amthor, 1984; Lötscher et al., 2004), we assumed respiration increased accordingly. The estimated respiration component shares over time are provided in Table S3 (Supplementary Material).

2.6.2. Drained peatland forest (DP)

Soil heterotrophic respiration in the post-thinning treated site was estimated using data from three trenched opaque automatic chambers located in the treated site. A detailed description of the automatic chamber system can be found in Laurila et al. (2021) and Koskinen et al. (2014). Calculation of hourly fluxes and flux filtering are described in Korkiakoski et al. (2020). To calculate annual heterotrophic respiration, gaps in the automatic chamber data were filled using a random forest model with conditional inference trees (Hothorn et al., 2006). The gap-filling procedure and model performance evaluation followed the approach described in detail by Rautakoski et al. (2024). Annual soil

heterotrophic respiration was then obtained by summing the gap-filled hourly fluxes, and the mean annual heterotrophic respiration from the three automatic chambers was used to represent the post-thinning treated site.

At the control site and the pre-thinning site, total forest-floor respiration was estimated using a combination of transparent automatic chambers ($n = 3$, located in control) and opaque manual chambers ($n = 15$), due to the lack of year-round heterotrophic respiration data. Winter respiration was estimated from automatic chambers and gap-filled using the previously described random forest model, with radiation included as a predictor. Flux processing followed the same procedure as for the post-thinning site. Total forest-floor respiration for the control and pre-thinning treated site was then estimated following a similar approach as in the UP, in which nighttime measurements from the chambers were used to model dark respiration using the [Lloyd and Taylor \(1994\)](#) equation excluding the RWC term, which was not significant (Eq. 4). The model was then used to estimate total forest-floor respiration on an hourly basis.

$$R = R0 * e^{E \left(\frac{1}{T0} - \frac{1}{T_{soil} - T1} \right)} \quad (4)$$

To determine the proportion of heterotrophic respiration from total forest-floor respiration, we used data from manual opaque chambers. Manual chamber measurements were done in three trenched and one non-trenched plot. The manual chamber description can be found in the [supplementary material S2](#). Fluxes were calculated using linear fitting ([Korkiakoski et al., 2020](#)). The heterotrophic-to-total respiration ratio derived from the manual measurements was then applied to the modeled dark respiration from the automatic chambers to estimate heterotrophic respiration for the pre-thinning treated site and the control. A common estimate was used for both pre-thinning treated and control sites.

2.7. Statistical analysis

All statistical analyses were performed using R (R Core Team, 2024). The data were checked for normality using the Shapiro–Wilk test. Normality was confirmed in the control sites and in the treated sites before thinning, but this assumption was violated in the treated sites after the thinning. Therefore, to test for differences in forest components between years and between control and treated sites, we used the non-parametric Mann–Whitney U-test. Statistical significance was assessed using a p-value threshold of 0.05. The uncertainty in the total carbon balance estimate was calculated by combining the standard errors (SE) of individual components using the standard error propagation method.

A sensitivity analysis was performed on the upland's heterotrophic respiration estimates to determine how variations in this key parameter affect the overall carbon budget, particularly NEP. Since heterotrophic respiration constitutes a major carbon flux in the system, changes in its assumed value can substantially impact NEP estimates. Specifically, we varied in the sensitivity analysis the response of tree root respiration to thinning by $\pm 10\%$ around the baseline assumption and assessed its consequent effect on heterotrophic respiration and the resulting changes in NEP.

3. Results

3.1. Carbon accumulation rate of the above and belowground forest components

In the UP, all components in the control and treated sites exhibited similar annual carbon accumulation rates before thinning ([Fig. 3A](#)). In the control, carbon accumulation rates remained stable throughout the monitoring period, except for stem carbon, which declined at the beginning of the period. In the treated site, after the growing season of

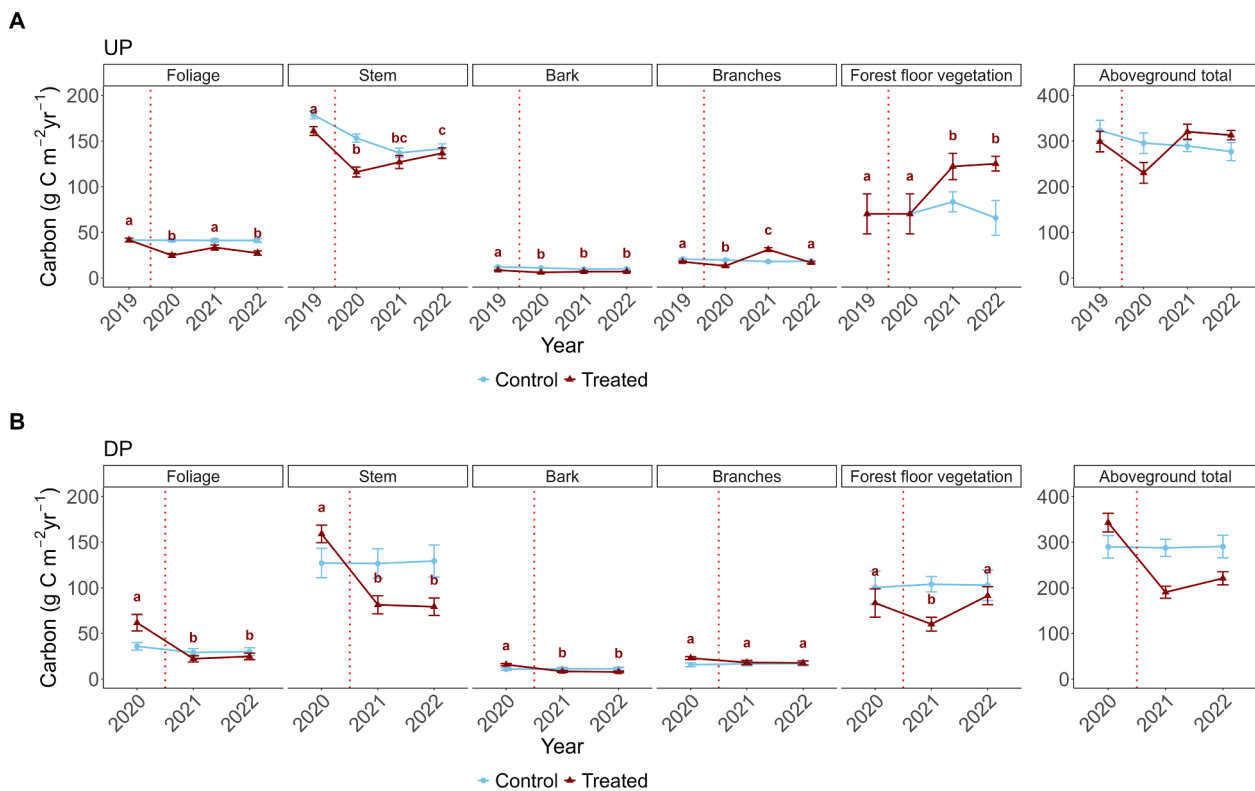


Fig. 3. Annual carbon accumulation rate in aboveground components of the forest and total aboveground carbon accumulation rate in (A) UP and (B) DP. The blue (red) lines represent carbon accumulation rate in the control (treated) site. Whiskers indicate standard errors. The dotted vertical line marks the time of the thinning. The right axis shows the values for total aboveground carbon accumulation rate. Letters indicate significant differences between years in the treated site (p-value < 0.05).

the thinning year, the carbon accumulation rate in the tree stem decreased from 161 to 116 g C m⁻² yr⁻¹, bark from 8.5 to 6 g C m⁻² yr⁻¹, foliage from 41.5 to 24.7 g C m⁻² yr⁻¹, and branches from 17.7 to 13.2 g C m⁻² yr⁻¹ (p value < 0.05), compared to pre-thinning levels. In the following year, foliage and branches showed a significant increase in carbon accumulation rate, to 33.4 g C m⁻² yr⁻¹ and 31.1 g C m⁻² yr⁻¹ respectively (p value < 0.05). Stems carbon accumulation rates showed a significant increase (p value < 0.05) two years after thinning.

The forest floor carbon accumulation rate increased from 70 to 122 g C m⁻² yr⁻¹ one year after the thinning and remained at a similar level in the subsequent year. Belowground components showed a decrease in carbon accumulation rates after thinning: stump carbon accumulation declined from 8.7 to 7.4 g C m⁻² yr⁻¹, and coarse root from 30.1 to 22.5 g C m⁻² yr⁻¹ (p value < 0.05). However post-thinning recovery was not significant until the second year after the intervention (Fig. 4A). Fine roots showed a significant decrease in their carbon accumulation rate. Total carbon accumulation rate decreased from 429.3 to 327.7 g C m⁻² yr⁻¹ after thinning and increased to 422.3 g C m⁻² yr⁻¹ the following year.

In the DP, the control and treated sites had some differences in their annual carbon accumulation rate before the thinning (Fig. 3B), with the treated site having a tree carbon stock that was 1.9 kg C m⁻² greater than that of the control. Before thinning, the stand basal area was higher in the treated site. After the growing season of the thinning year, tree carbon stock in the treated site decreased from 9.8 kg C m⁻² to 4.5 kg C m⁻². Following this, during the first growing season after thinning, the annual carbon accumulation rate significantly declined (p value < 0.05) in foliage (from 61.7 to 22.3 g C m⁻² yr⁻¹), stem (from 159 to 81.4 g C m⁻² yr⁻¹), branches (from 22.9 to 18.1 g C m⁻² yr⁻¹) and bark (from 15.8 to 8.3 g C m⁻² yr⁻¹) (Fig. 3B). These components maintained stable accumulation rates thereafter, with no significant changes observed in the subsequent year. The carbon accumulation rate in forest floor vegetation declined in the first year after thinning, then increased to a level higher than before thinning (from 60 to 91 g C m⁻² yr⁻¹) (Fig. 3B). The tree stump (from 10.8 to 6.4 g C m⁻² yr⁻¹) and coarse roots (from 31.3 to 19.6 g C m⁻² yr⁻¹) showed a significant reduction (p value <

0.05) in annual carbon accumulation following thinning (Fig. 4B), with rates stabilizing in the subsequent year.

3.2. Total carbon emissions

During the thinning, 2.2 ± 0.2 kg C m⁻² was removed from the forest in the UP and 3.9 ± 0.4 kg C m⁻² in the DP, in the form of stem and bark. Residues such as branches and foliage remained on-site. Table 2 summarizes the partitioning of all the components of the aboveground decomposition.

Soil respiration in the UP was partitioned according to the assumptions described in the Methods Section (2.6.1). Fig. 5 presents the contributions of heterotrophic respiration, tree root respiration, and forest floor vegetation respiration (including mycorrhizae). Tree root respiration decreased following thinning, while heterotrophic and forest floor vegetation respiration increased.

In the UP, the total carbon loss through decomposition of

Table 2

Annual carbon emissions from aboveground decomposition (g C m⁻² yr⁻¹) of different components in thinned sites at UP and DP. The 2019 data in UP and the 2020 data in DP are pre-thinning conditions.

Parameter	UP				DP		
	2019	2020	2021	2022	2020	2021	2022
Deadwood (Mg ha ⁻¹)	7.4 ± 1.9	7.1 ± 1.8	6.8 ± 1.8	6.6 ± 1.7	25.0 ± 7.6	24.0 ± 7.3	23.0 ± 7.0
Branch residue (Mg ha ⁻¹)	–	54.0 ± 3.5	47.5 ± 3.1	41.7 ± 2.7	–	63.8 ± 9.2	39.5 ± 5.7
Foliage residue (Mg ha ⁻¹)	–	35.4 ± 4.0	30.4 ± 3.5	14.1 ± 1.6	–	57.3 ± 5.5	57.7 ± 5.6
Stump (Mg ha ⁻¹)	–	4.5 ± 0.3	4.4 ± 0.3	4.2 ± 0.3	–	11.7 ± 1.0	11.2 ± 0.9
Total (Mg ha ⁻¹)	7.4 ± 1.9	101.0 ± 5.6	89.1 ± 5.0	66.6 ± 3.6	25.0 ± 7.6	156.7 ± 13.0	131.4 ± 10.7

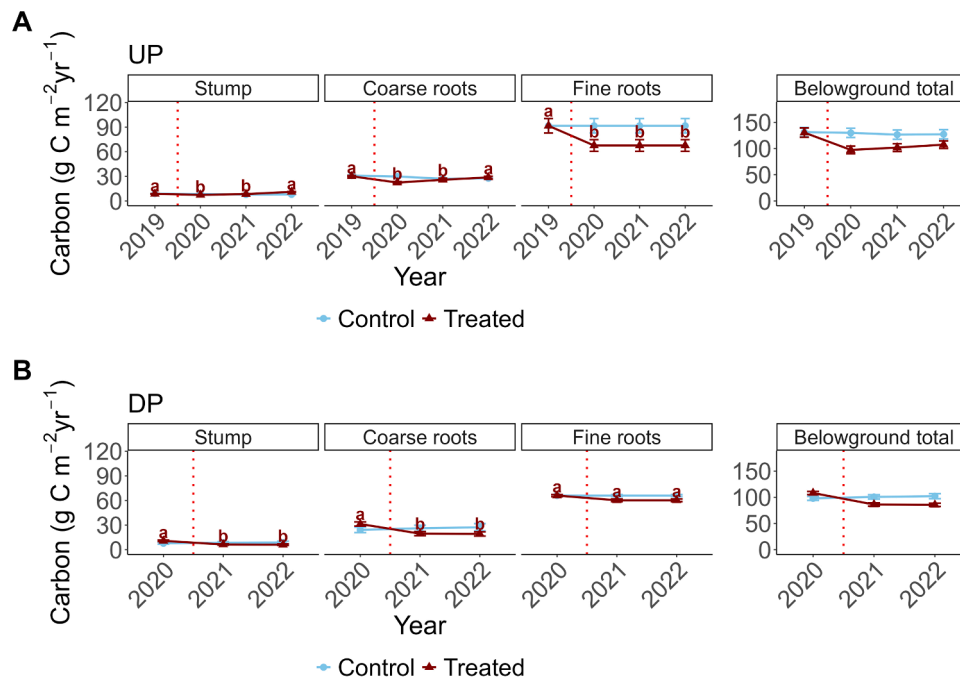


Fig. 4. Annual carbon accumulation rate in belowground components of the forest and total belowground carbon accumulation rate in (A) UP and (B) DP. The blue (red) lines represent carbon accumulation rate in the control (treated) site. Whiskers indicate standard errors. The dotted vertical line marks the time of the thinning. The right axis shows the values for total belowground carbon accumulation rate. Letters indicate significant differences between years in the treated site (p-value < 0.05).

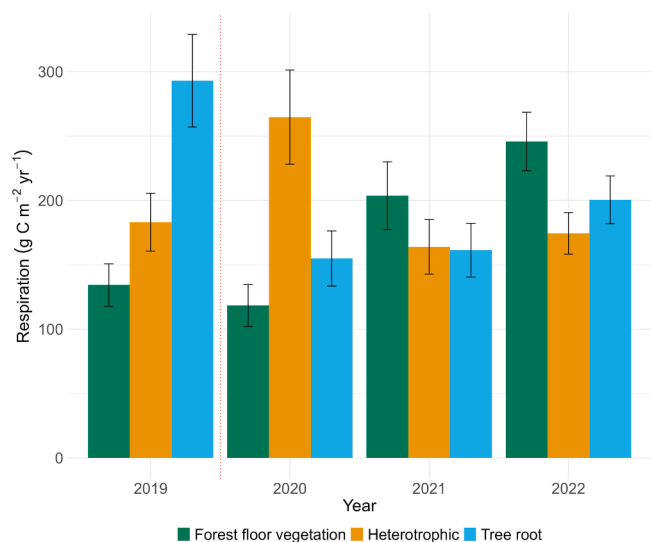


Fig. 5. Partitioning of soil respiration in the UP treated site, showing heterotrophic, tree root, and forest floor vegetation respiration. Whiskers indicate standard errors. The dotted red line represents the time of thinning.

aboveground components in the thinned site significantly increased to $101 \text{ g C m}^{-2} \text{ yr}^{-1}$ after the thinning and remained high the following years (Table 2; Fig. 6A). In the DP, carbon loss through the decomposition significantly (p value < 0.05) increased to $157 \text{ g C m}^{-2} \text{ yr}^{-1}$ and remained high the year after the thinning (Table 2; Fig. 6B). No significant changes in heterotrophic respiration were detected in either forest after thinning.

3.3. Carbon budget

The net ecosystem production (NEP) was calculated by balancing the previously estimated carbon accumulation rate and carbon losses of the forest. Carbon loss due to leaching was estimated using fixed values of 5 for the UP (Rätty et al., 2020) and $16 \text{ g C m}^{-2} \text{ yr}^{-1}$ for DP (Rantakari et al., 2010), assuming no effect of thinning on leaching.

In the UP, NEP was positive (net carbon sink) before thinning, it

turns negative (net carbon source) in the year following the thinning, and returned to net carbon sink the subsequent years, although it remained lower than pre-thinning levels (Fig. 7A). The control site remained a stable net carbon sink throughout the study period (Fig. 7B).

In the DP, NEP was negative in both the control and thinned sites before thinning (Fig. 8C,D). NEP values in the thinned site decreased further in the year after thinning and remained negative the following year. NEP in the control remained negative throughout the study period.

4. Discussion

Our results suggest that the short-term effects of thinning on forest carbon balance may vary between upland and drained peatland conditions. In this study, we observed contrasting responses at the two sites: the UP shifted relatively quickly back toward carbon uptake after an initial decline, whereas the DP remained a stronger carbon source one year after thinning. Although these findings are based on single sites, they highlight the potential role of site-specific factors, such as soil properties and microclimate, in shaping carbon dynamics following management interventions like thinning. Given the large extent and high carbon storage capacity of boreal forests (Kurz et al., 2013; Deluca and Boisvenue 2012; Clemmensen et al., 2013), Even small changes in carbon balance at the local level could matter when looking at the bigger picture, which highlights the need to manage forests in ways that take both the site conditions and climate goals into account.

We first hypothesized (H1) that the carbon accumulation rate of the forest components will initially decrease following thinning but gradually start to recover to pre-thinning levels over time. Our results support this hypothesis as we observed a decline in the carbon accumulation rate of all forest components in both forest type caused by thinning, with a recovery in the subsequent year. This fast recovery was largely driven by forest floor vegetation. The UP substantially recovered (74 %) to pre-thinning levels two years after thinning, whereas the DP had not reached pre-thinning levels one year after thinning. The UP outcome was expected, as thinning removes both carbon stocks and reduces the potential for carbon sequestration from thinned trees. However, thinning also typically benefits the remaining trees by reducing competition and can enhance forest productivity in the short term. This pattern was observed in our Scots pine dominated UP, where aboveground carbon accumulation rate increased already a year after the thinning and

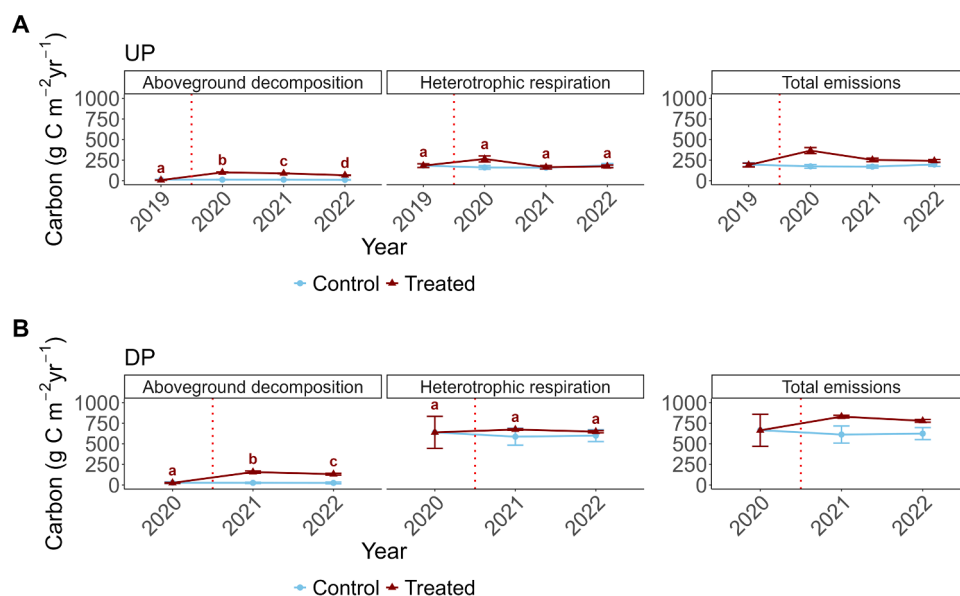


Fig. 6. Annual carbon emissions in (A) UP and (B) DP. The blue (red) lines represent carbon fluxes in the control (treated) site. Whiskers indicate standard errors. The dotted line represents the time of the thinning. The right axis shows the values for total carbon emissions. Letters indicate significant differences between years in the treated site (p -value < 0.05).

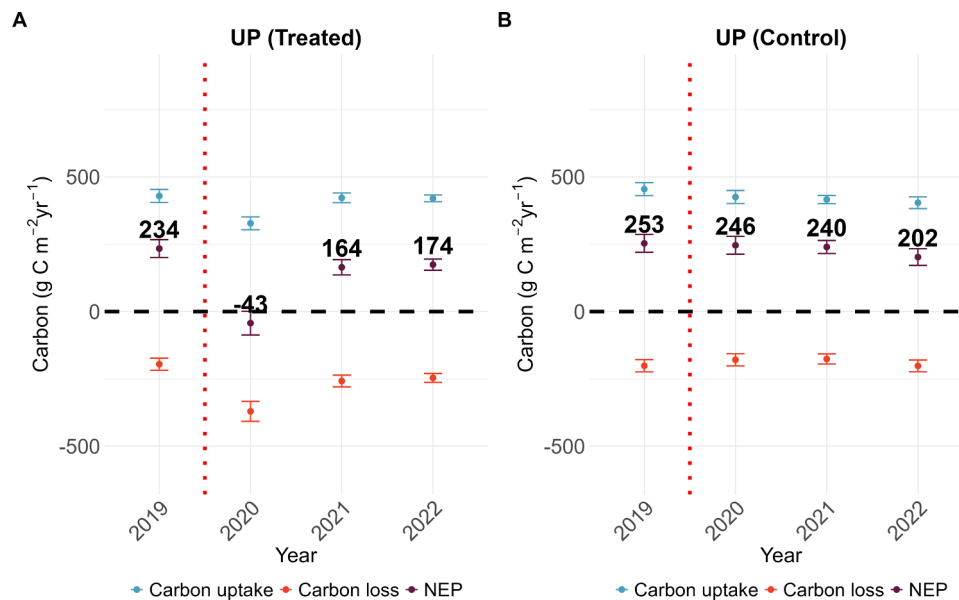


Fig. 7. Carbon budget showing carbon accumulation rate, carbon losses, and net ecosystem production (NEP) in the UP: (A) treated site and (B) control. Positive values indicate carbon sink in the forest, while negative values indicate carbon release to the atmosphere. For NEP, positive values represent a net carbon sink to the ecosystem, whereas negative values indicate that the forest is a net carbon source to the atmosphere. Whiskers indicate standard errors.

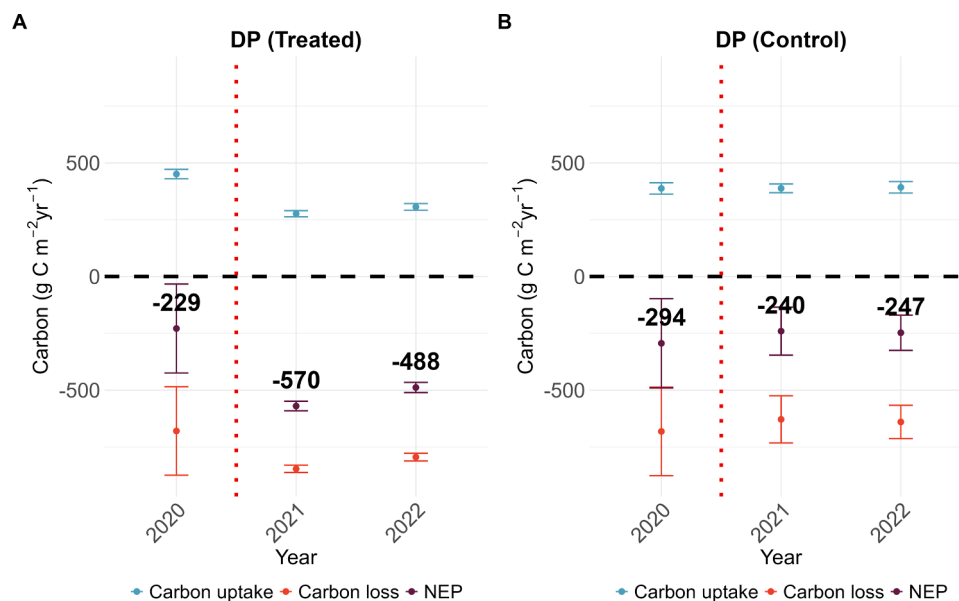


Fig. 8. Carbon budget showing carbon accumulation rate, carbon losses, and net ecosystem production (NEP) in the DP: (A) treated site and (B) control. Positive values indicate carbon sink in the forest, while negative values indicate carbon release to the atmosphere. For NEP, positive values represent a net carbon sink to the ecosystem, whereas negative values indicate that the forest is a net carbon source to the atmosphere. Whiskers indicate standard errors.

further after two years as we observed significantly higher carbon accumulation rates than immediately after the thinning. This is consistent with the findings of Mäkinen and Isomäki (2004a), who reported that annual diameter growth increased after thinning in Scots pine-dominated stands on mineral soils in southern and central Finland, which would be expected to enhance carbon sequestration in tree biomass.

In contrast, in the Norway spruce dominated DP, aboveground tree carbon accumulation rate did not show a clear increase after thinning. This may reflect a slower recovery or a tendency for trees to initially allocate resources to belowground biomass. While Mäkinen and Isomäki (2004b) observed increased annual diameter growth following thinning in Norway spruce-dominated forests in southern and central Finland,

their findings were based on stands growing on mineral soils. The lack of a strong aboveground response in our drained peatland site may be due to the absence of a distinct suppressed cohort that would typically benefit from release by thinning, along with drier soil conditions in the middle of the growing season following thinning. We expected an increase in the water table levels after thinning, which could have also limited the growth response; however, this was not observed, most likely due to effective drainage. Another contributing factor may be the species-specific allocation patterns of Norway spruce. In drained peatlands, Norway spruce trees may respond to post-thinning conditions without delay but initially allocate increased growth to root systems (Lehtonen et al., 2023, 2025b). However, this belowground allocation change would not be fully captured by the allometric equations used in

this study, which may partly explain the absence of a clear belowground response. Similarly, in the UP, we did not observe a significant recovery in coarse root biomass. While the impact of thinning on root carbon allocation remains understudied, Joseph et al. (2020) reported that increases in soil moisture following dry conditions can significantly influence belowground allocation patterns in Scots pine. Such conditions could potentially occur after thinning, but this was not the case in our study. Although thinning slightly increased soil moisture, pre-thinning conditions were never sufficiently dry to trigger a notable belowground response, as reported by Joseph et al. (2020). This suggests that Scots pine in the UP allocated more carbon to the aboveground components which would explain its significant increase post-thinning. Additionally, both forests experienced a significant increase in carbon accumulation rate by forest floor vegetation after thinning likely due to increased light availability from canopy opening. Forest floor vegetation is particularly efficient in light utilization (Kolari et al., 2006), so the combination of increased light and soil resources likely explains the rapid rise in the productivity of forest floor vegetation.

The results also support our second hypothesis (H2) that post-thinning, carbon emissions increase due to enhanced biomass decomposition of harvest residues. This is primarily driven by the large quantities of branches and foliage left at the site as residue, as only tree stems are removed during thinning. A similar increase in residue emissions following thinning has been reported by Vesala et al. (2005). In this study, we assume that all the carbon in the woody debris is eventually lost as CO₂ through decomposition. While some of this carbon may be stabilized into the soil, studies have found no significant effects of thinning on soil carbon stocks (Roth et al., 2023; Zhang et al., 2018). However, it is important to note that the stumps and coarse roots of the removed trees will remain on the forest floor for a longer period, continuing to contribute to the soil carbon stock. Removing harvest residues—whether through whole-tree harvesting or other methods—may seem like a potential option to reduce carbon emissions on site. However, such practices can reduce the carbon storage in the soil (Repo et al., 2015) and disrupt nutrient balances in the forest (Hendrickson et al., 1989; Duchesne and Houle, 2008; Ouimet et al., 2021), potentially decreasing stand growth. Furthermore, the removed biomass might be used for bioenergy (Natural Resources Institute Finland (Luke), 2024), leading to an immediate release of its stored carbon into the atmosphere, resulting in a sharp increase in carbon emissions (Mäkipää et al. 2015; Repo et al., 2012). We found no significant differences in soil heterotrophic respiration following thinning. However, previous studies, such as Noormets et al. (2012) and Lei et al. (2018), have reported post-thinning increases in heterotrophic respiration, often linked to soil warming and accelerated decomposition of organic matter. In our study, the absence of a statistically significant increase may reflect the influence of methodological assumptions and uncertainties in partitioning soil respiration, which are discussed in more detail later.

We found that thinning initially reduced the carbon sink strength in both forests, followed by a pronounced recovery in the UP one year after thinning, and a more modest recovery in the DP. These findings partially support our hypothesis (H3) that the slower NEP recovery in the DP, compared to the UP, was due to higher heterotrophic respiration and decomposition of harvest residues. However, lower increase in above-ground carbon accumulation after thinning also contributed to the slower recovery. Although the two forest types showed different carbon sequestration recovery patterns, their NEP remained below pre-thinning levels two years after thinning in the UP and one year after thinning in the DP. Aun et al. (2021a) reported a similar trend in an upland pine forest in Estonia, where within two years after thinning, young and middle-aged stands had not fully recovered to control levels. In the UP, our NEP results align with the net ecosystem exchange (NEE) dynamics described by Aslan et al. (2024) with eddy covariance method. They found that the UP (Hyytiälä) became a carbon source after thinning but recovered to a carbon sink in a year. Additionally, our NEP estimation for the UP control and pre-thinning align with the NEE values reported

by Ilvesniemi et al. (2009) for the same UP over a 10-year period (ranging from 141 to 247 g C m⁻² yr⁻¹). While the NEE has not yet been studied in DP (Ränskälänkorpi) based on eddy-covariance, Korhonen et al. (2023) investigated the effects of thinning in Lettosuo, a similar drained peatland forest in southern Finland, using the eddy covariance method. Their findings indicate that, before thinning, the forest alternated between years of being a carbon source and a weak sink, with a six-year pre-thinning average classifying it as a weak net carbon source. In the DP, we only collected data from one year before thinning, which suggests that it was a stronger net carbon source than Lettosuo, likely due to site differences. Furthermore, Korhonen et al. (2023) found that Lettosuo remained a strong net carbon source for three years post-thinning before transitioning into a carbon sink. Our results indicate a reduction in the net carbon source in the DP one year after thinning; further study spanning a few years after thinning would be needed to confirm this transition. These findings show that, in this particular case, the carbon balance recovered more quickly in the UP than in the DP. While upland forests typically show a relatively fast NEP recovery after thinning and approach the pre-thinning levels in few years (Aslan et al., 2024; Aun et al., 2021a), drained peatlands show a different pattern, in which nutrient-rich peat soils that were carbon sources before thinning tend to remain so in the short term after thinning, as shown by Korhonen et al. (2023). However, a more robust conclusion regarding sites comparisons would require replication across additional sites, as this study included only one upland and one drained peatland forest.

Our findings show a rapid recovery of NEP in the UP after thinning. However, if we look into the tree carbon stocks, which is the main carbon storage affected by thinning, we can see that the recovery of the carbon storage is not as pronounced. The tree pre-thinning carbon storage of the treated site was 8.5 kg C m⁻², after the thinning the stock was reduced to 6.1 kg C m⁻², and after two years it recovered to 6.5 kg C m⁻² (without considering the decomposition of the roots of the removed trees). This means that a large amount of carbon that was stored in the ecosystem was removed in the thinning and although the NEP has substantially recovered in two years after the thinning, it would take over 13 years for the carbon stock to recover to pre-thinning levels at the current growth rate. The situation in the DP is similar. After thinning the tree carbon stock in the treated site was reduced from 9.8 kg C m⁻² to 5.5 kg C m⁻² and then recovered to 5.6 kg C m⁻² a year after thinning. Since we only have the results of the stand response a year after the thinning, it is difficult to predict the carbon stock recovery. At the current growth rate, recovery would take approximately 33 years, although it is reasonable to assume that growth will eventually accelerate. However, at this stage, it is not possible to predict the extent or duration of any future increase. It is also important to note that in our NEP calculations, we did not account for the carbon removed in stem biomass during thinning, as this carbon is not directly released into the atmosphere within the forest. This reduction in forest carbon stocks is not necessarily negative for the climate in case the harvested carbon remains stored in wood products for several years—and in the case of building materials, even for several decades (Profft et al., 2009; Aktas and Bilec, 2012). However, a remarkable part of these removed trees may be typically used for bioenergy (Karttunen et al., 2016; Seppo et al., 2013), which releases the stored carbon back into the atmosphere, representing a significant carbon source. Furthermore, data from the Natural Resources Institute of Finland (Luke) *Flows of Wood Material* statistics database (<https://www.luke.fi/en/statistics>) show that in 2023, about 60 % of all harvested forest products were used for energy. Therefore, to comprehensively assess the carbon balance in these forests, it would be essential to account for the final use and lifespan of the wood products from the thinning, as these factors ultimately determine the long-term carbon balance outcome of the thinning procedure.

In this study, several assumptions were necessary, particularly in estimating heterotrophic respiration in the UP after thinning. We assumed that tree removal reduced autotrophic root respiration while

increasing heterotrophic respiration due to soil organic matter and fine root decomposition, shifting the soil respiration balance. Similar shifts have been reported in other systems: Noormets et al. (2012) observed increased heterotrophic respiration following thinning in a *Pinus taeda* plantation, while Vesala et al. (2005) suggested a comparable shift after finding no significant change in total soil respiration post-thinning. Additionally, girdling experiments in Scots pine—effectively halting photosynthate supply to roots—led to sustained reductions in root respiration for up to two years (Högberg et al., 2001; Singh et al., 2003) potentially indicating a shift in the total soil respiration partitioning. In our analysis, we assumed that the shift in the ratio between autotrophic and heterotrophic respiration was linked to the quantity of root biomass remaining after thinning, increasing the relative contribution of heterotrophic respiration at the expense of tree root respiration. However, partitioning soil respiration remains uncertain due to variability in root decay, microbial dynamics, and soil moisture (Auffret et al., 2016; Kelsey et al., 2012; Karhu et al., 2014). Additional uncertainty arose from heterotrophic respiration estimates in the peatland, mostly due to differences in measurement techniques (automatic vs. manual chambers) that may have introduced bias in the pre-thinning heterotrophic respiration estimations. Due to limited data, we also assumed that forest floor vegetation in the treated sites was similar to that in the control site at the year of the thinning and one year before. Leaching values were based on literature rather than site-specific measurements, which introduces some uncertainty but is unlikely to have had a major influence on the main findings. Additionally, the measurement of fine root growth has limitations, as explained by Lukac (2012). As a result, we considered the growth of fine roots as the average over a two-year period measurement. Similar to forest floor vegetation, we assumed that the control site was representative of the treated site conditions prior to thinning for both study sites. These assumptions, however, may not fully capture the differences between the treated and control site conditions.

Overall, the methodological variability may have affected the precision of our carbon balance estimates and combined with those inherent in the eddy covariance approach (Campioli et al., 2016), might explain the discrepancy in the absolute NEP values compared to Aslan et al. (2024) in the UP. Despite of this, the dynamics aligned, suggesting that the overall trends remain robust. These uncertainties highlight the need for more detailed empirical data on soil respiration partitioning and decomposition rates in disturbed systems. Long-term, multi-site studies would improve understanding and reduce uncertainty in post-thinning carbon balance assessments. Nevertheless, our findings are at ecosystem scale consistent with previous studies, including some conducted in the same area, and the sensitivity analysis of the heterotrophic respiration assumptions (Table S4 in supplementary data) increases our confidence in the reliability of our general conclusions.

Using biometric estimates alongside soil chamber data offers a detailed view of management effects on forest carbon components and helps to clarify how different carbon fluxes respond to disturbances like thinning. Our results highlight the importance of considering site-specific responses when evaluating thinning effects on the carbon budgets in forests. While thinning appears to have a transient effect on carbon sequestration in upland forests, its impact on peatland forests may be more prolonged due to slower tree biomass recovery and persistent soil emissions. This is particularly important given that one-third of Finland's forest cover is classified as peatland. To better understand and manage these dynamics, more long-term, replicated studies are needed on thinning effects in drained peatland forests. In particular, future research should address (1) the role of microclimatic changes and drainage conditions in post-thinning carbon responses, (2) the long-term recovery trajectories of net ecosystem productivity and tree carbon stocks in peatland stands, and (3) the temporal dynamics of above- and belowground carbon allocation in Norway spruce after thinning. These knowledge gaps are especially critical considering the extensive area and high carbon storage potential of managed peatland forests in boreal regions.

Although thinning affects forest carbon accumulation rates and stocks, it remains a sustainable forest management practice. When implemented with consideration of stand age at the time of harvest, thinning can even enhance the long-term carbon stock in the forest (Pukkala, 2017). In addition, by reducing competition-based mortality, thinning can lower the amount of decomposing deadwood on the forest floor and help maintain the stand's carbon sequestration level (Neumann and Hasenauer, 2021; Pretzsch et al., 2023). Thinning also increases forest resilience to wildfires and drought, helping to prevent damage from severe climate-induced disturbances that could otherwise lead to substantial carbon losses (Seidl et al., 2014; Kasischke et al., 1995). Taken together, the potential for long-term carbon stock enhancement, reduced deadwood-related emissions, and improved resilience to climate-induced disturbances highlight how well-planned thinning can balance supporting timber production and sustaining long-term carbon sequestration in forest ecosystems.

Declaration of Competing Interest

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

Acknowledgements

This work was supported by Research Council of Finland funding for BiBiFe project “Biogeochemical and biophysical feedbacks from forest harvesting to climate change” (grant nr. 324259), for ForClimate project “Managing Forests for Climate Change Mitigation” (347782, European Union – NextGenerationEU funding instrument), InterEarth university profiling project (353218), ROBAST Academy Fellow project “The role of bark in tree survival under drought stress” (355142) and Atmosphere and Climate Competence Center Flagship (357902), Research Council of Finland funding (grant nr. 312571 and grant nr. 357263), CANEMURE (EU Life IP), HoliSoils (H2020 grant no 101000289) and Ministry of Agriculture and Forestry, Finland (Catch the Carbon Programme, Advances in Soil Information). YS was supported by Research Council of Finland funding (grant nr. 323843 and 357263).

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.foreco.2025.123024.

Data Availability

The data is available in Zenodo (10.5281/zenodo.15646898)

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