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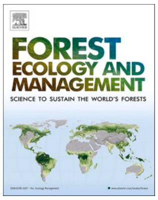

Initial impacts of forest management on forest floor greenhouse gas fluxes in hemiboreal coniferous forests on drained nutrient-rich organic soils

Valters Samariks [a,*], Raija Laiho [b], Jyrki Jauhiainen [b], Aldis Butlers [a], Dovilė Čiuldienė [c], Andis Lazdiņš [a], Muhammad Kamil-Sardar [d], Thomas Schindler [d], Kaido Soosaar [d], Egidijus Vigricas [c], Ieva Līcīte [a]

[a] Latvian State Forest Research Institute "Silava", Salaspils LV-2169, Latvia
[b] Natural Resources Institute Finland (Luke), Latokartanonkaari 9, Helsinki FI-00790, Finland
[c] Institute of Forestry, Lithuanian Research Centre for Agriculture and Forestry, Instituto al. 1, Akademija, Kėdainiai LT-58344, Lithuania
[d] Institute of Ecology & Earth Sciences, University of Tartu, 46 Vanemuise, Tartu EST-51014 Estonia

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ABSTRACT

As greenhouse gas (GHG) concentrations in the atmosphere continue to rise, legal frameworks are implemented internationally, nationally, and regionally to address ways to mitigate climate change. These frameworks emphasize the role of forests in sequestering carbon dioxide and the challenges in managing forests effectively amidst climate change and increasing timber demand. There is thus an urgent need for comprehensive understanding of soil GHG fluxes to plan forest-sector climate change mitigation strategies effectively. This study aims to evaluate the initial effects of various forest management types: regeneration cut, partial cut, commercial thinning, and a combination of commercial thinning and wood ash fertilization, on GHG exchange of forest floor (soil, litter layer, roots, above-ground parts of ground vegetation) in hemiboreal conifer-dominated forests with drained nutrient-rich organic soils. Forest management significantly influenced forest floor GHG fluxes through differences in soil temperature and soil water-table level (WTL). Forest floor total respiration that describes overall soil activity responded variably to different employed management types, while in intact forests (Control sites) it decreased with increasing stand age. Regeneration cut resulted in CH_4 emissions due to higher WTL and reduced evapotranspiration, while other management types maintained the forest floor as a CH_4 sink. N_2O flux varied across the management types; however, the flux remained minor in all cases. Overall, partial cut exhibited the lowest GHG flux increase. While our study covers initial responses, longer-term studies are needed to fully evaluate the management impacts on drained nutrient-rich organic soils that have high potential for contributing to GHG fluxes.

1. Introduction

Greenhouse gas (GHG; CO_2, CH_4, and N_2O) concentrations in the atmosphere continue to rise. This impacts the climate system and forms a global environmental issue (IPCC, 2014; 2022). Hence, legal frameworks are developed at global (UNFCCC, Kyoto Protocol, Paris Agreement) (UNFCCC, 1997; 2015), European (EU Green Deal) (Fetting, 2020), and national ((LCS) Latvia Climate Strategy, 2019) levels to mitigate GHG emissions and to adapt to climate change. Forests remove carbon dioxide (CO_2) from the atmosphere by increasing the tree biomass carbon (C) pool, part of which is transferred to the soil through litterfall (Friedlingstein et al., 2022); however, soil C net loss may occur from drained organic forest soils (peatlands) (Wüst-Galley et al., 2016). The estimated global forest C stock is 662 gigatons, of which almost half is in soil organic matter (FAO, 2020). Forest ecosystems of boreal and hemiboreal biomes form significant terrestrial carbon pools, especially in forest soils, particularly organic soils, which retain more C than overstories (Schulze et al., 1999; Ķēniņa et al., 2022). Overstory tree biomass C stock and C sequestration capacity is mainly dependent on tree species, stand age, and site fertility (Vogel and Gower, 1998). Natural disturbances and forest management practices can alter the C stocks. Moreover, soil GHG emissions and soil C stocks strongly depend on forest management types (Mäkipää et al., 2023). The uncertainty regarding changes in the soil C pool poses significant challenges for

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* Corresponding author.
E-mail address: valters.samariks@silava.lv (V. Samariks).

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climate change mitigation in forest land. This uncertainty partly arises from the forest management types implemented and is expected to intensify as the impacts of climate change unfold. Furthermore, the demand for timber and wood products will likely continue to increase in the future, emphasizing the need for effective forest management practices that can match the timber production demand without compromising climate change mitigation on one hand and adaptation on the other.

Currently, the dominant management method in the hemiboreal region is even-aged forestry (Duncker et al., 2012). The final harvest is conducted through regeneration cut, which is followed by the soil scarification to promote the growth and vitality for the subsequently planted seedlings. Typical management practices include pre-commercial thinning, along with one to three additional thinning operations prior the final harvest. Typical rotation periods for coniferous (Norway spruce (*Picea abies* (L.) H. Karst.), Scots pine (*Pinus sylvestris* L.)) stands vary from 80 to 100 years. Between 1960 and 1980, extensive forest drainage initiatives were undertaken in the Baltic states to enhance tree growth and increase the productivity of forests growing in areas with water-table levels naturally close to soil surface, i.e., typically organic peat soils (Paavilainen and Päivänen, 1995; Šņore, 2004). Typically, the drainage systems are also periodically reconditioned to maintain their drainage function supporting oxygen availability for the root systems and the desired tree growth rates (Nieminen et al., 2018). However, the drawdown of the soil water-table level (WTL) leads to elevated CO_2 fluxes and potentially soil C loss due to the greater exposure of peat to aerobic decomposition processes in the formed oxic conditions, particularly in nutrient-rich organic soils (Ojanen et al., 2010). Accordingly, it has been suggested that alternative management options, such as partial harvests, should be applied in forests on organic soils to reduce their GHG emissions (Nieminen et al., 2018; Korkiakoski et al., 2023). However, there is currently only limited data and understanding on the impacts of different management types on the GHG fluxes from these sites (Jauhiainen et al., 2023). Consequently, there is an urgent need to deepen our understanding of the functions of drained organic forest soils and their interaction with the atmosphere and neighboring ecosystems.

The CO_2 balance in forest ecosystems depends on the ecosystem's potential to store C in biomass and in soil, where C inputs to soil with litter become subject to C losses caused by decomposition (Franko and Ruehlmann, 2018). Forest management activities affect both the soil and vegetation, therefore potentially increasing forest floor CO_2 respiration due to decomposition of ground vegetation and cutting residues, often combined with increased soil temperature that increases soil organic matter decomposition (Korkiakoski et al., 2019). Generally, drained forest soils may act as either sinks of (i.e., gaining) or sources of (i.e., losing) CO_2, depending on site characteristics and climate (Ojanen et al., 2010, Jauhiainen et al., 2023, Butlers et al., 2025). Methane (CH_4) emission and uptake rates, in turn, can alter with changes in WTL (Ojanen et al., 2010; Tong et al., 2022). Forested wetlands may be significant sources of soil CH_4 emissions if their soils are predominantly under anoxic conditions (Matthews and Fung, 1987), but forests on drained organic soils may form a small soil CH_4 sink as the oxic surface soil layer allows microbial oxidation of CH_4 (Le Mer and Roger, 2001). An additional important aspect is CH_4 emissions from drainage ditches, which can considerably contribute to the CH_4 budget, even though they occupy a rather small fraction of a forest ecosystem (Peacock et al., 2021; Rissanen et al., 2023). Nitrous oxide (N_2O) emissions are generally higher from drained than undrained organic forest soils, especially in nutrient-rich sites (Martikainen et al., 1993; Minkkinen et al., 2020). Typically, N_2O fluxes are linked to soil nitrogen (N) stock, and a low soil C:N ratio may result in high N_2O emissions (Martikainen et al., 1993; Ojanen et al., 2010). Moreover, N_2O fluxes from organic soils are reported to increase with rising soil NO_3^- and are linked to soil moisture and soil temperature change (Pärn et al., 2018).

After regeneration cut, organic forest soils have been reported to be sources of CO_2 and CH_4 emissions due to increased soil temperature and WTL, as all forest cover is removed and evapotranspiration reduced (Korkiakoski et al., 2019). Partial cut and thinning, in turn, have been reported as management types that result in small or even insignificant effects on soil GHG fluxes due to low soil disturbance (Shabaga et al., 2015; Korkiakoski et al., 2020, 2023). N_2O emissions usually increase after regeneration cut as mineral N is released from logging residues after management (Korkiakoski et al., 2019). Fertilization by wood ash has been a common practice in Nordic countries; however, less practiced in the Baltic states. Effects of fertilization vary depending on soil type, forest type, time since fertilization, and the quality and quantity of wood ash applied (Mäkipää et al., 2023). Soil CO_2 fluxes from organic soils may be dependent on time since fertilization: in short-term ($<$ 5 years) fluxes have not increased but may do so in the longer term, especially in sites with high soil nitrogen (N) concentrations (Mäkipää et al., 2023). Emissions of CH_4 are unlikely to rise because of the increase in forest growth, as the increased evapotranspiration of trees lowers the WTL (Mäkipää et al., 2023). Nitrous oxide fluxes do not appear to depend on wood ash fertilization in boreal peatland forests (Ojanen et al., 2019; Mäkipää et al. 2023). Notable knowledge obtained about organic soils in the boreal region is valuable for forming an understanding of potential outcomes, but is likely not directly transferable to regions differing by, e.g., climate and land-use history. Therefore, a thorough understanding of GHG fluxes is needed in the hemiboreal region, where organic soils are also common, to plan forest-sector climate change mitigation measures.

The aim of this study is to (1) quantitatively assess the mean daily, monthly, and annual GHG fluxes from intact forest stands (no management during the study) and stands managed with regeneration cuts, partial cuts, commercial thinning, or commercial thinning with wood ash fertilization, (2) compare the fluxes of the managed stands to intact stands, and the fluxes of different-aged intact forests, and (3) evaluate how water-table level and temperature vary among managed versus intact stands, and how they are related to the growing-season GHG fluxes, for which the most extensive data are available. We hypothesize that (i) forest management will result in elevated forest floor CO_2 respiration, (ii) forest management will increase forest floor CH_4 emissions due to reduced evapotranspiration caused by tree removal, and (iii) forest management will increase forest floor N_2O fluxes.

2. Materials and methods

2.1. Study sites

The 15 study sites, all located across the territory of Latvia in the hemiboreal forest zone (Ahti et al., 1968), consisted of drained nutrient-rich organic forest soils (peatlands) with varying forest management types (Table 1; Table S1, Fig. 1 A). All sites were situated on organic soils with peat layer ranging from 30 cm to more than 2 m, corresponding to *Oxalidosa turf. mel.* and *Myrtillosa turf. mel.* forest types, both characterized as nutrient-rich soils based on stand productivity and ground vegetation composition (Bušs, 1997).

The climate in the region is characterized as temperate but strongly affected by the Baltic Sea. The mean annual precipitation sum (30-year mean from 1991 to 2020) is 686 mm, and the mean annual air temperature is + 6.8 °C. The coldest month is February with a mean air temperature of −3.1 °C, and the warmest month is July, with a mean of + 17.8 °C ((LEGMC) Latvian Environment, 2020). Mean monthly variation of air temperature and monthly precipitation sum of monitoring years was compared to the 30-year mean air temperature and precipitation in Latvia to evaluate interannual differences (Fig. S1). All monitoring years had similar or slightly higher monthly mean air temperatures compared to the 30-year mean observation, with the largest differences observed for winter month air temperatures between monitoring years (Fig. S1 A). On average, all monitoring years were warmer than the 30-year mean, with mean annual air temperature 9.3 °C, 9.8 °C, 8.1 °C, and 8.5 °C from 2019 to 2022, respectively. Mean

Table 1
Description of the study sites on drained nutrient-rich organic forest soils before (pre) and after (post) management operations. Values are expressed as mean ± 95 % confidence interval. See Table S1 for site-level information of monitoring periods. DBH – Diameter at breast height; WTL – soil water-table level (downwards from the soil surface) for the whole monitoring period.

Site ID	Management type	Management status	Age	DBH, cm	Tree height, m	Basal area, m^2ha^{-1}	Trees per ha, n	Growing stock, m^3	WTL, cm	Soil temperature at 10 cm, °C
409–225–21	Control	Intact forest	36	19.7	18.1	32	1320	201.1	-28.6	8.3
409–402–10	Control	Intact forest	34	18.2	16.2	25	620	305.1	-33.8	8.7
409–474–21	Control	Intact forest	39	19.5	17.9	20	720	174.0	-88.3	8.8
410–213–8	Control	Intact forest	2	2.0	0.5	NA	2000	NA	-40.4	8.8
410–218–3	Control	Intact forest	21	21.0	20.9	26	760	277.4	-41.5	8.4
410–218–16	Control	Intact forest	70	23.0	23.0	30	320	351.5	-74.3	8.8
E_Valle	Control	Intact forest	161	25.4	24.7	32	320	630.3	-24.7	10.3
E_Zalve	Control	Intact forest	157	32.0	30.0	29	680	354.3	-46.3	11.4
E_Auce	Control	Intact forest	171	43.2	33.8	42	1320	400.7	-66.0	11.1
Group mean	**Control**	**Intact forest**	**77 ± 49**	**24.9 ± 5.8**	**23.0 ± 4.2**	**29.4 ± 4.3**	**727 ± 256**	**335.6 ± 96.0**	**-23.0 ± 4.2**	**9.4 ± 0.9**
LVC112	RC	Pre-management	162	10.2	10.2	21	1820	176.0	-30.7	6.4
LVC112	RC	Post-management	0	NA	NA	NA	3000	NA	-17.3	8.4
LVC116	RC	Pre-management	141	14.3	14.0	34	1593	337.7	-31.1	5.5
LVC116	RC	Post-management	0	NA	NA	NA	2000	NA	-7.5	8.8
Group mean	**RC**	**Pre-management**	**152**	**12.3**	**12.1**	**27.5**	**1707**	**257**	**-21.7**	**7.3**
LVC308	PC	Pre-management	141	25.3	23.2	36	647	451.7	-50.4	5.6
LVC308	PC	Post-management	141	NA	NA	NA	NA	NA	-31.0	8.2
LVC313	PC	Pre-management	141	13.6	13.3	39	1893	398.0	-52.5	6.2
LVC313	PC	Post-management	141	NA	NA	NA	NA	NA	-35.4	10.2
Group mean	**PC**	**Pre-management**	**141**	**19.5**	**18.3**	**37.5**	**1270**	**424.9**	**-42.3**	**7.6**
LVC113	CT	Pre-management	48	19.9	20.6	42	1360	455.7	-69.9	-0.4
LVC113	CT	Post-management	48	NA	NA	NA	NA	NA	-69.6	6.8
Group mean	**CT**	**Pre-management**	**48**	**19.9**	**20.6**	**42**	**1360**	**455.7**	**-69.8**	**3.2**
LVC307	CT+WA	Pre-management	48	23.2	21.9	40	920	450.0	-56.0	1.1
LVC307	CT+WA	Post-management	48	NA	NA	NA	NA	NA	-56.1	7.6
Group mean	**CT+WA**	**Pre-management**	**48**	**23.2**	**21.9**	**40**	**920**	**450.0**	**-56.1**	**4.4**

Note: NA – information not available. The old-growth control stands (E_Valle, E_Zalve, E_Auce) correspond to E. Buchwald's (2005) old-growth forest category 'n6', representing prolonged rotation.

monthly precipitation for monitoring years was highly variable, and the mean annual precipitation sum from 2019 to 2022 was 575 mm, 498 mm, 582 mm, and 583 mm, respectively (Fig. S1 B); thus, monitoring years were on average drier than the 30-year mean, and the differences in mean annual air temperature and mean monthly precipitation were statistically non-significant between monitoring years.

The tree stands of the sites were dominated by Norway spruce (*Picea abies* (L.) H. Karst.), except for the partial and regeneration cut sites, where Scots pine (*Pinus sylvestris* L.) was a co-dominant species (Table 1). The ground vegetation in the Oxalidosa type sites was characterized by forbs such as *Oxalis acetosella Dryopteris* spp., and *Calamagrostis* spp., and moss layer dominated by *Pleurozium schreberi*, *Dicranum* spp., *Hylocomium splendens*, and *Rhytidiadelphus triquetrus*. In the Myrtillosa type sites, patches of dwarf shrubs such as *Vaccinium myrtillus*, *V. vitis-idaea*, and herbs such as *Dryopteris* spp., *Rubus saxatilis*, *Maianthemum bifolium* alternated with patches of forest mosses such as *Pleurozium schreberi*, *Dicranum* spp., *Hylocomium splendens*, *Rhytidiadelphus triquetrus* or *Sphagnum* mosses. The described site ground vegetation characteristics correspond to their pre-treatment conditions.

Nine study sites experienced no management during the study period and were considered as control sites, labeled as "intact forest", representing young to old-growth forest stands (2–171 years old). Another six study sites were classified as intact forests during initial assessments ("pre-management"). After baseline measurements, these sites underwent one of four different forest management types, and data collection ("post-management"). The lengths of monitoring periods varied among sites as described in Table S1. Management types included regeneration cut (RC; 2 sites) of spruce and pine dominated forests, partial cut (PC; 2 sites) of spruce and pine dominated forests, commercial thinning (CT; 1 site) of spruce forest, and commercial thinning combined with wood ash fertilization (CT+WA; 1 site) of spruce forest (Table 1; Table S1).

Regeneration cut included clear-felling, soil scarification with mounding, and tree planting. Partial cut included removal felling of both small and large trees, removal of understory trees, and harvesting the stand only partially to allow remaining trees to grow larger and create openings for natural regeneration. Partial cutting was performed to achieve a minimal basal area ± 2 m^2 ha^{-1} based on common forest management practices in Latvia. Minimal basal area is determined state law based on dominant tree species and mean stand height; in this case, it ranged from 10 to 12 m^2 ha^{-1}. Commercial thinning included stand thinning based on common forest management practices in Latvia with thinning intensity of ca. 30 % of total basal area by removing deciduous, understory and suppressed trees. In commercial thinning combined with wood ash fertilization, fertilization was conducted four months after thinning with a dose of 5 t ha^{-1}.

2.2. GHG sampling and flux calculation

At each site, a plot was reserved for data collection, which took place in 1–3 sub-plots prepared for monitoring. At each of the Control sites (intact forest), one sub-plot (R= 12.62 m, A= 500 m^2) with five GHG

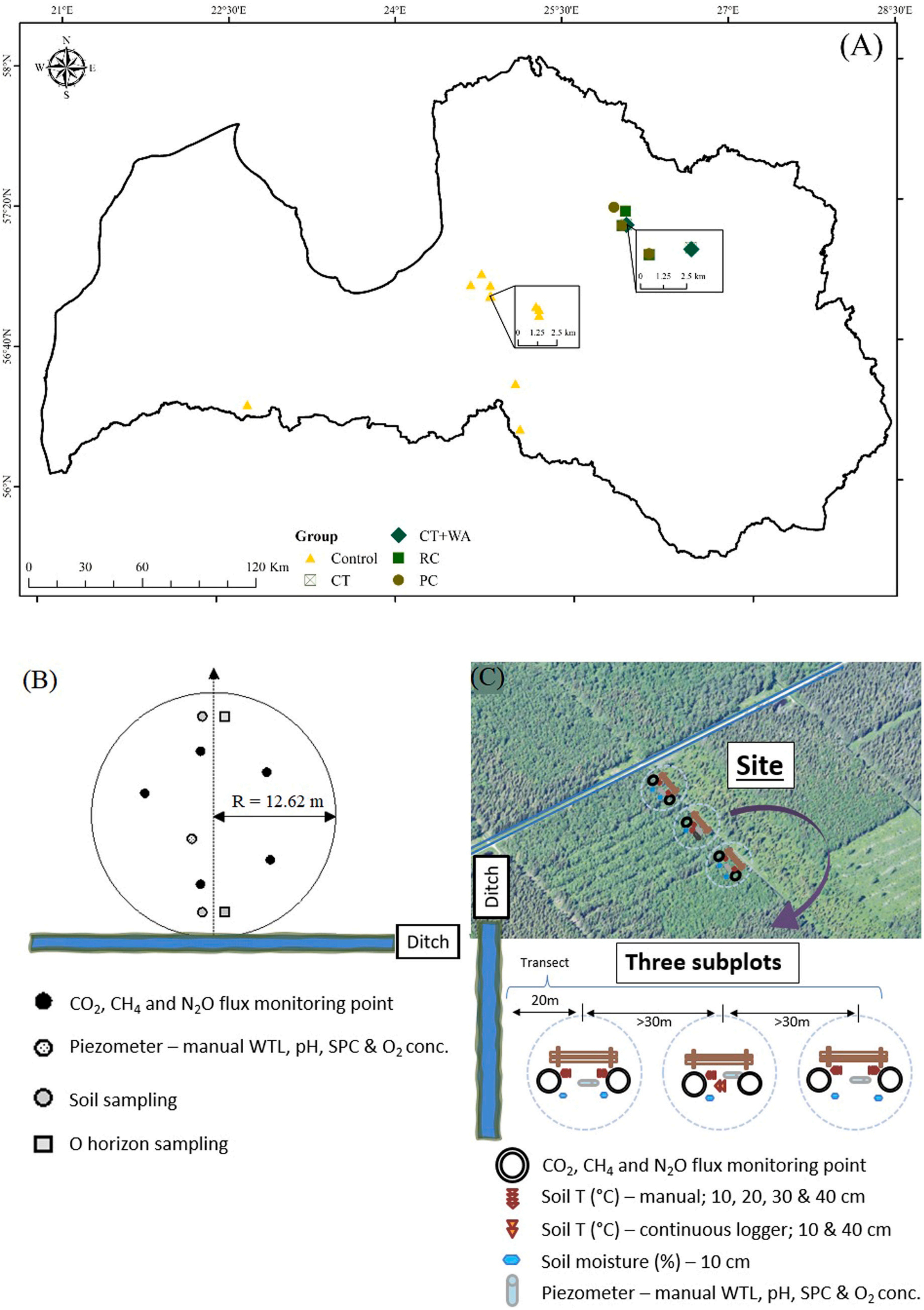

Fig. 1. The location of study sites in the territory of Latvia (A) and schematic design of one sub-plot established for data collection in Control sites (B) and that of three sub-plots in sites with management (C).

monitoring points was set in a representative place in the middle of the site (Fig. 1 B). At the other sites, three sub-plots, each with two GHG monitoring points, were established along a straight line (transect) through the study sites (Fig. 1 C), with a minimum of 15 m from the forest edge.

The measurements of soil GHG (CO_2, CH_4, and N_2O) fluxes were carried out using the dark chamber method (Hutchinson and Livingston, 1993). The chamber had two parts: a conical PVC chamber (height 30 cm, diameter 50 cm, volume 65 liters) and a stationary measurement collar inserted into the soil before the monitoring campaign to minimize disturbance to the ground vegetation. The sampled air included soil-atmosphere GHG fluxes, emissions from the ground vegetation present on the monitored soil surface, as well as from tree roots extending there. In this study, the monitored CO_2 fluxes are thus referred to as forest floor respiration (ER_{ff}), encompassing both autotrophic and heterotrophic components of soil respiration. It should be noted that the respiration flux thus does not represent soil CO_2 balance, while the CH_4, and N_2O represent the respective soil balances.

To collect air samples for GHG analysis, the chamber was sealed airtight onto the collar, and a syringe connected via tubing was attached to the chamber. A series of four air samples were collected from the closed chamber at 20-minute intervals over a 60-minute incubation period using pre-evacuated 50 ml bottles. Subsequently, the air samples were transported to the laboratory of the Department of Geography at the University of Tartu, where the concentrations of GHGs (CO_2, CH_4, N_2O) were determined by gas chromatography. The gas chromatograph Shimadzu GC-2015 was equipped with an electron capture detector, a flame ionization detector, and a Loftfield autosampler (Loftfield et al., 1997). The GHG samples were taken once per month during the daytime, between 10:00 and 14:00 local time, a period that generally corresponds to average daily temperature (Parkin and Venterea, 2010). During this time window, the monitoring points of the site were measured in random order. The monitoring periods for each site are listed in Table S1.

GHG fluxes were calculated by utilizing the linear regression slope of gas concentration changes over time within the closed chamber. Measurements were excluded if the coefficient of determination (R^2) of the regression fell below 0.9 ($p < 0.01$), unless the difference between the highest and lowest CO_2 concentrations in the chamber was within the method's uncertainty, typically observed during non-vegetation periods, in which case those measurements were retained. Overall, less than 5 % of the data was excluded. Only measurements that met the quality criteria were used to assess the linear regression slope coefficient used in flux calculation.

$$flux = \frac{M \times P \times V \times slope}{R \times T \times A \times 1000} \tag{1}$$

where *flux* is the soil GHG flux (mg GHG m^{-2} h^{-1}); *M* is the molar mass of GHG (g mol^{-1}); *R* is a universal gas constant (m^3 Pa K^{-1} mol^{-1}); *P* is the assumption of air pressure inside the chamber (101.300 Pa); *T*—air temperature (K); *V*—chamber volume (0.063 m^3); *slope*—rate of change of the hourly GHG concentration (ppm GHG h^{-1}) inside the chamber; *A*—collar area (0.1995 m^2).

2.3. Soil sampling and analyses

Soil samples were collected with a volumetric 100 m^3 probe at 0–10, 10–20, 20–30, and 30–40 cm depths with two replications in each sub-plot before the start of GHG measurements. Bulk density, pH_{KCl}, and concentrations of total carbon (C_{tot}), nitrogen (N_{tot}), phosphorus (P_{tot}), potassium (K_{tot}), calcium (Ca_{tot}), and magnesium (Mg_{tot}) were determined for all samples (Table 2) according to ISO standards (Table S3).

2.4. Measurements of environmental parameters during field campaigns

Environmental parameters were measured during the gas sampling campaigns once per month. The WTL was monitored manually as a distance downwards from the soil surface to water level at each sub-plot from a mesh-coated, perforated PVC pipe inserted in the soil down to a depth of 200 cm. Soil temperature was measured at 10 cm depth with Comet Pt1000 temperature sensors (COMET SYSTEM, Czech Republic). Water temperature, conductivity, and pH were measured with a YSI ProDSS multiparameter analyzer (YSI /Xylem Inc., Yellow Springs, OH, USA) from the WTL monitoring pipe water. Monthly precipitation sums were obtained from Latvian Environment, Geology and Meteorology Centre (LEGMC) meteorological stations (Fig. S1)

2.5. Statistical analyses

At site level the obtained mean hourly (mg m^{-2} h^{-1}) fluxes from each chamber were multiplied with the hours in a day into mean daily fluxes (g m^{-2} d^{-1} for CO_2, and mg m^{-2} d^{-1} CH_4, N_2O). Mean monthly fluxes (g m^{-2} d^{-1} for CO_2, and mg m^{-2} d^{-1} CH_4, N_2O) were obtained by averaging mean daily fluxes of each month. These mean monthly fluxes of specific months were then transformed to total monthly fluxes kg ha^{-1} $month^{-1}$ by multiplying them with the number of days in each specific month. Total annual CO_2, $CH_{4,}$ and N_2O fluxes (t ha^{-1} $year^{-1}$) were calculated by summing up the total monthly fluxes in each year. Positive flux values denote emission from the forest floor to the atmosphere, and negative values denote uptake from the atmosphere. The total annual GHG flux as CO_2 equivalents (CO2eq t ha^{-1} $year^{-1}$) was calculated by multiplying each annual GHG with the global warming potential (GWP; CO2 = 1; CH4 = 27; N2O = 273) on a 100-year time horizon, according to the IPCC guidelines (IPCC, 2022).

Due to the limited pre-management measurements on both sites managed with thinning (Table S1), we used two approaches for examining the management impacts: site-level comparison of the pre- and post-management fluxes, where applicable, and comparison of the post-management fluxes to a larger group of non-managed (control) sites which also included pre-management measurements. We used mean daily fluxes from April to October, labeled as the "snow-free period", with consistent data coverage across sites, for most analyses. The Shapiro-Wilk test was used to assess the normality of dependent variables. Logarithmic transformation was performed to meet the normal distribution for non-parametric data. Management impact was assessed as the differences between GHG fluxes of treatment means (i.e. the comparison of the post-management fluxes to the larger group of Control (intact forest) sites, and site-level comparison of the pre- and post-management fluxes of snow-free period) with the Tukey's HSD test.

Table 2

Soil characteristics of the study sites; for sites that were actively managed during the study, they represent the pre-management conditions. The values are means ± 95 % confidence intervals for a 0–40 cm soil layer.

Site management	Bulk density, kg m^{-3}	pH_{KCl}	C_{tot}, t ha^{-1}	N_{tot}, g kg^{-1}	P_{tot}, g kg^{-1}	K_{tot}, g kg^{-1}	Ca_{tot}, g kg^{-1}	Mg_{tot}, g kg^{-1}
Control	239.8 ± 40.8	4.3 ± 0.2	712.8 ± 3.9	4.1 ± 0.3	0.2 ± 0.02	0.1 ± 0.02	3.4 ± 0.4	0.3 ± 0.1
CT	826.6 ± 258.6	5.0 ± 0.3	479.9 ± 3.9	3.3 ± 1.0	0.4 ± 0.1	0.6 ± 0.2	4.3 ± 1.3	0.7 ± 0.2
CT+WA	216.0 ± 10.3	4.9 ± 0.2	848.3 ± 4.6	6.2 ± 0.3	0.3 ± 0.1	0.1 ± 0.03	6.1 ± 0.5	0.3 ± 0.1
RC	203.6 ± 104.9	3.4 ± 0.5	327.5 ± 2.1	3.2 ± 0.7	0.1 ± 0.02	0.1 ± 0.03	1.6 ± 0.7	0.1 ± 0.03
PC	308.3 ± 152.9	3.4 ± 0.3	438.2 ± 3.4	2.7 ± 0.4	0.1 ± 0.02	0.2 ± 0.2	1.3 ± 0.5	0.2 ± 0.1

Hypotheses were tested using Tukey's HSD test and ANOVA. All analyses were performed with a 95 % confidence interval.

Linear mixed effect models were developed for analyzing which factors were most strongly related to the variation in the GHG fluxes. We used site-level mean daily flux for each analyzed GHG separately and added multiple fixed effects as long as it improved the model. Fixed effect variables of the final models were selected based on variable significance and model improvements indicated by the lowest Akaike information criterion (AIC) value. Site and measurement month were used as random effects in all linear mixed effect models. All analyses were performed with RStudio (R Core Team, 2021).

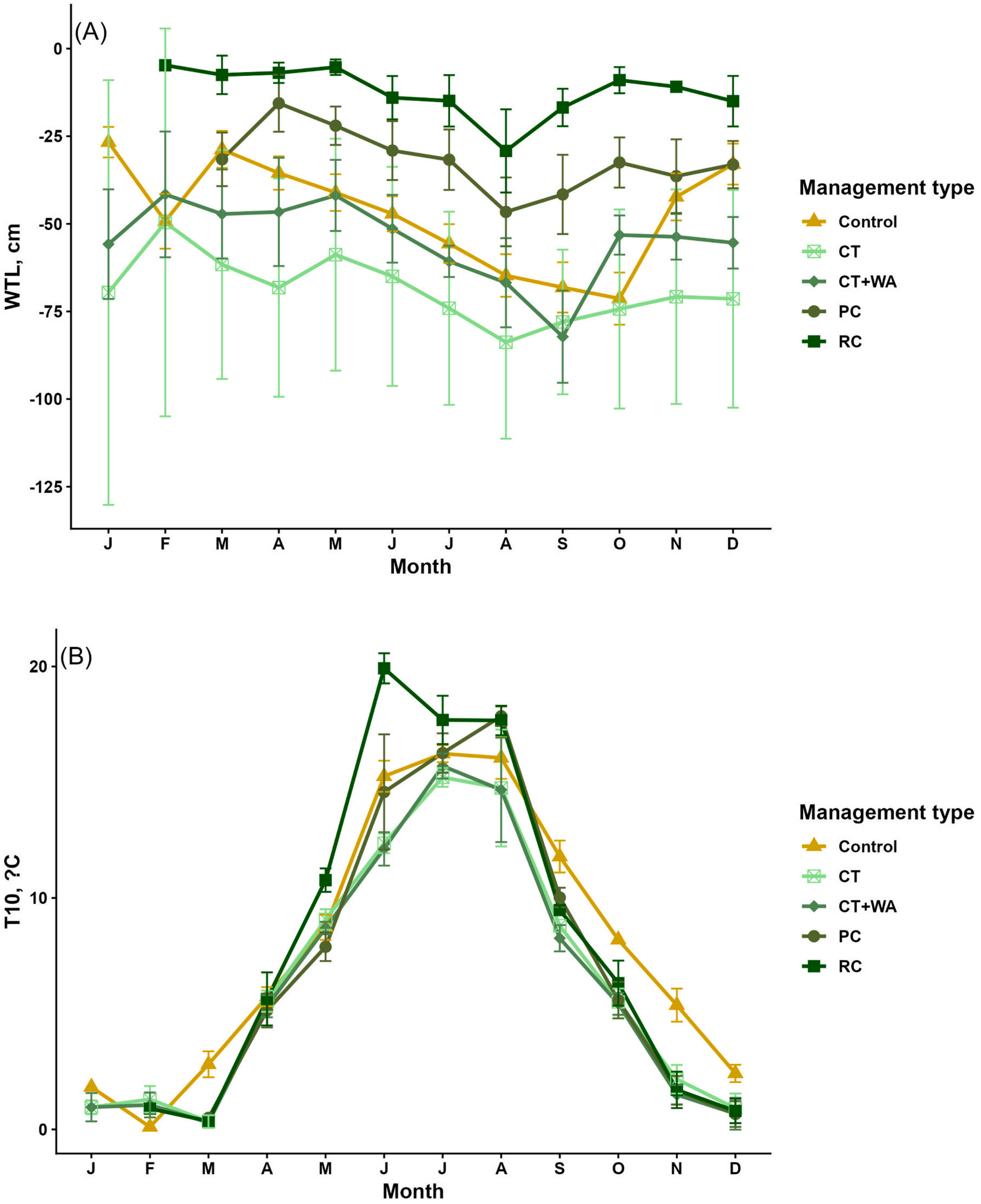

Fig. 2. The distribution of mean monthly soil water-table level (A; WTL, cm below soil surface) and soil temperature at 10 cm depth (B; T10, °C) at study sites with different forest management types. The symbols denote means, and the whiskers 95 % confidence intervals. Management types: Control – forest that was not managed during the study period and sites prior management; CT – commercial thinning; PC – partial cut; RC – regeneration cut; CT+WA – commercial thinning combined with wood ash fertilization. Note that the monitoring periods were not the same for all sites (Table S1), see Fig. S1 for monthly weather patterns for the monitoring years and Fig. S2 for mean WTL and T 10 for the snow-free period in pre- and post-treatment periods.

3. Results

3.1. Management impact on soil temperature and water table level (WTL)

Mean soil WTL varied significantly among the management types ($p < 0.05$; Fig. 2A). The highest mean WTL values were observed for the RC (-12.36 ± 1.98 cm), followed by the PC (-32.96 ± 2.86 cm). Control sites and the CT+WA management exhibited similar WTLs −47.53 ± 1.99 cm and −56.09 ± 3.27 cm, respectively, but the CT showed the lowest values (-69.60 ± 7.65 cm; Fig. 2A). Moreover, forest management significantly increased WTL in the RC and PC compared to pre-management state (Fig. S2A).

Mean annual soil temperature at 10 cm depth was similar in all management types ($p > 0.05$), but some differences could be observed between management types at certain months (Fig. 2B). Comparison of mean soil temperature for the snow-free period prior and post-management application exhibited increase in soil temperature for the RC post-management; however such differences were not observed for

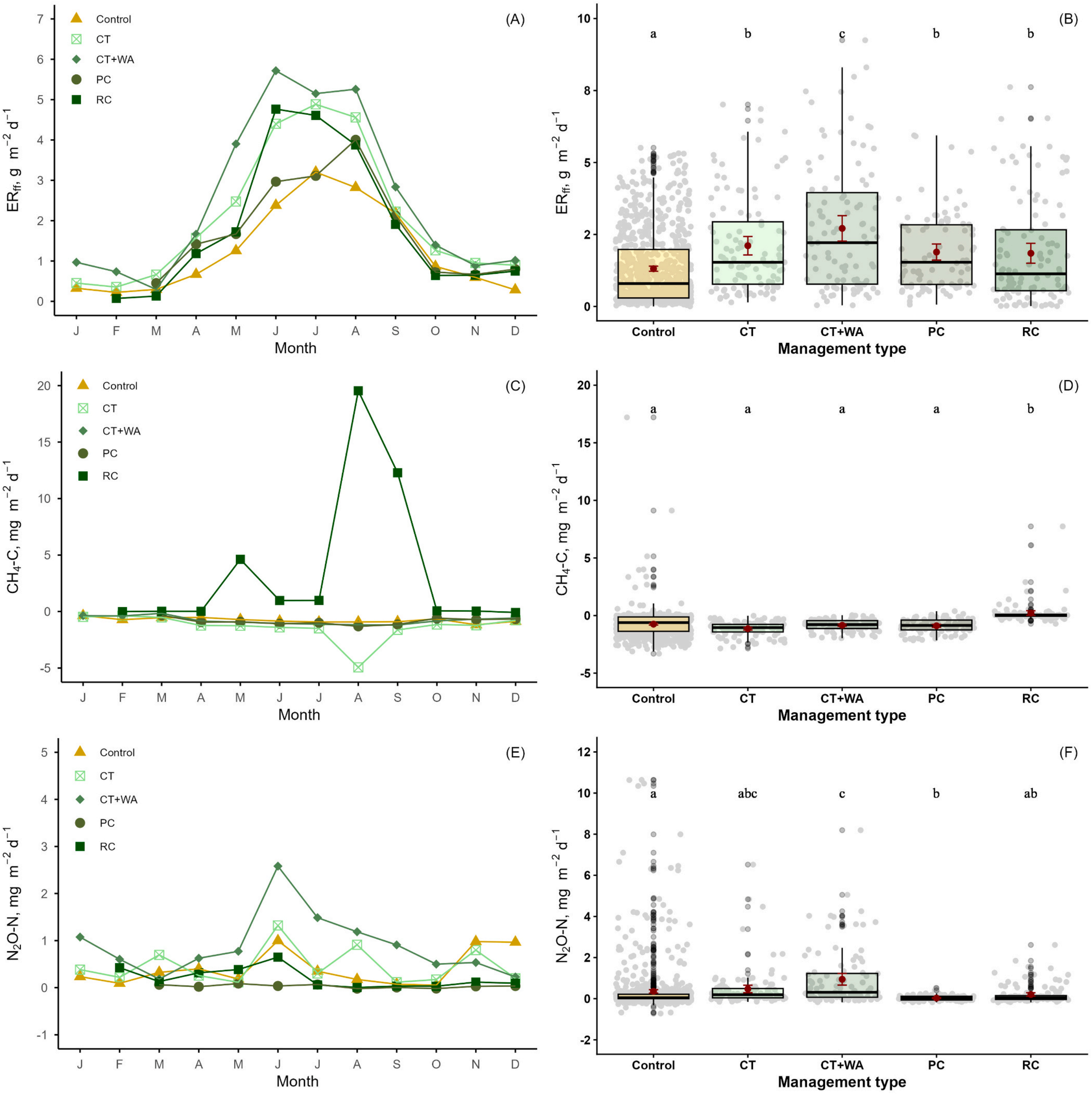

Fig. 3. Mean monthly fluxes (A, C, E) and mean daily fluxes (B, D, F) of GHGs (ER_{ff}, CH_4, N_2O) at study sites with different forest management types. Abbreviations: Control –forest that was not managed during the study period and sites prior management; CT – commercial thinning; PC – partial cut; RC – regeneration cut; CT+WA – commercial thinning combined with wood ash fertilization. In figures B, D, and F gray dots show single data points; red dots indicate *daily* mean values ± 95 % confidence intervals; the box plots represent *daily* median values and the 25 % and 75 % percentiles; the whiskers indicate the maximum and minimum values; and the black dots represent outliers. The letters in each figure with bars indicate statistically significant differences in treatment means at the $p < 0.05$ level. Note that ER_{ff} involves both heterotrophic respiration from organic matter decomposition and autotrophic respiration from above-ground biomass of ground vegetation and belowground biomass of ground vegetation and trees. Monitoring periods were not the same for all sites (Table S1).

the PC (Fig. S2B). Moreover, for the CT and the CT+WA mean soil temperature for the snow-free period post-management was significantly lower than for the Control sites (Fig. S2B).

3.2. Mean daily and the snow-free period fluxes, and the impacts of forest management

3.2.1. ER_{ff}

The Control sites showed the lowest mean daily ER_{ff}, (1.31 ± 0.09 CO_2-C g m^{-2} d^{-1}) while the ER_{ff} in all forest sites post-management was significantly ($p < 0.05$) higher (Fig. 3 B). The highest mean daily ER_{ff} was 2.80 ± 0.47 CO_2-C g m^{-2} d^{-1} observed in the CT+WA sites being significantly ($p < 0.05$) higher than that of all other evaluated management types. Mean daily ER_{ff} was similar between the CT, PC and RC. In thinned (CT) sites ER_{ff} was 2.21 ± 0.37 CO_2-C g m^{-2} d^{-1}, after partial cut (PC) 1.89 ± 0.28 CO_2-C g m^{-2} d^{-1}, and after regeneration cut (RC) 1.85 ± 0.35 CO_2-C g m^{-2} d^{-1}. The mean monthly ER_{ff} (g C m^{-2} d^{-1}) followed a clear seasonal trend. Higher fluxes were observed during the vegetation season from April to October and lower fluxes during cold season (Fig. 3 A).

Mean daily ER_{ff} for the snow-free period of Control sites ranged from 0.96 ± 0.11–2.88 ± 0.64 CO_2-C g m^{-2} d^{-1} and from 1.67 ± 0.25–2.42 ± 0.43 CO_2-C g m^{-2} d^{-1} in pre-management sites. Mean daily ER_{ff} for the snow-free period of post-management sites ranged from 2.12 ± 0.38–3.79 ± 0.63 CO_2-C g m^{-2} d^{-1}, with the lowest mean values observed for PC site, but the highest for CT+WA. Mean daily ER_{ff} for the snow-free period did not significantly differ between pre-treatment and post-treatment in RC and PC sites (Fig. 4 A), but post-treatment Mean daily ER_{ff} for the snow-free period of the RC was significantly higher compared to Control sites. Overall, the CT post-management Mean daily ER_{ff} for the snow-free period was significantly higher ($p < 0.05$) than control and any pre-management ER_{ff}, however, the CT+WA post-management Mean daily ER_{ff} for the snow-free period was significantly higher ($p < 0.05$) than control and all evaluated management (pre- and post-management) types, except the post-management CT (Fig. 4 A).

The grouped post-management vs. control comparison indicated significantly higher mean daily ER_{ff} for all managed sites relative to Control sites (Fig. 3 B). In contrast, in the site-level pre– vs. post-management comparison mean daily ER_{ff} for the snow-free period of the PC sites was similar compared to the Control sites (Fig. 4 A). The relative ranking of post-management treatments also differed: in the grouped comparison CT+WA showed the highest ER_{ff}, significantly exceeding all other treatments but CT, PC, and RC were statistically similar to each other (Fig. 3B), whereas in the site-level approach, CT+WA remained highest, significantly exceeding all other treatments except CT (Fig. 4A). There was a significant negative linear relationship between mean daily ER_{ff} for the snow-free period and stand age at the Control sites ($p < 0.01$; Fig. S3 A).

3.2.2. CH_4

Negative mean daily CH_4 fluxes (methane uptake) were observed for most management types. Mean daily CH_4 flux in the Control sites was −0.75 ± 0.08 CH_4-C mg m^{-2} d^{-1}. Further, soil CH_4 uptake from the atmosphere was observed for the PC, CT, and CT+WA management types with −0.89 ± 0.13 CH_4-C mg m^{-2} d^{-1}, −1.451 ± 0.41 CH_4-C mg m^{-2} d^{-1}, and −0.84 ± 0.10 CH_4-C mg m^{-2} d^{-1}, respectively (Fig. 3 D), and the differences between Control, PC, CT, and CT+WA were non-significant. However, the post-management RC sites acted as CH_4 emitters for the majority of measurement period, except in February and December with mean daily CH_4 emissions 3.50 ± 4.16 CH_4-C mg m^{-2} d^{-1}, showing significantly higher fluxes ($p < 0.001$) compared to all other management types and Control. The mean monthly CH_4 fluxes did not follow a clear seasonal trend (Fig. 3 C).

Mean daily CH_4 fluxes for the snow-free period of all sites ranged from −2.09 ± 0.17 CH_4-C mg m^{-2} d^{-1} to 11.14 ± 13.16 CH_4-C mg m^{-2} d^{-1} and were similar between Control sites and all evaluated management (pre- and post-management) types, except for the RC sites post-management mean daily CH_4 flux for the snow-free period, which was significantly ($p < 0.05$) higher (Fig. 4 B). In contrast, no significant relationships were found between stand age and mean daily CH_4-C fluxes for the snow-free period ($p > 0.05$; Fig. S3 B).

3.2.3. N_2O

Mean daily N_2O flux for the Control sites was 0.38 ± 0.09 N_2O-N mg m^{-2} d^{-1} and showed relatively higher mean monthly fluxes in November and December. The highest mean daily N_2O-N fluxes were observed for the CT+WA treatment (0.94 ± 0.28 N_2O-N mg $m^{-2}d^{-1}$). The CT+WA differed significantly from Control, PC and RC, but not from the CT, where the mean daily N_2O-N flux was 0.47 ± 0.18 N_2O-N mg $m^{-2}d^{-1}$. The mean daily N_2O flux in the RC was 0.20 ± 0.09 N_2O-N mg $m^{-2}d^{-1}$. The lowest mean daily N_2O-N flux during the entire monitoring period was observed for the PC treatment (0.03 ± 0.03 N_2O-N mg $m^{-2}d^{-1}$), which was significantly lower than the mean for the Control sites (Fig. 3 F). Mean monthly N_2O fluxes did not reveal a clear seasonal trend, apart from most management types showing peak N_2O fluxes in June (Fig. 3 E).

Mean daily N_2O fluxes for the snow-free period ranged from −0.006 ± 0.03 N_2O-N mg m^{-2} d^{-1} to 2.03 ± 1.01 N_2O-N mg m^{-2} d^{-1}. Mean daily N_2O fluxes for the snow-free period were similar between the Control, pre-management PC sites, and post-management PC, RC, and CT (Fig. 4 C). Significantly, higher mean daily N_2O fluxes for the snow-free period were observed in post-management CT+WA sites, and pre-management RC sites, in comparison to Control, pre-management PC, post-management PC, and RC sites. While post-management CT+WA remained the treatment with the highest mean daily N_2O fluxes for the snow-free period, a significant decrease in mean daily N_2O fluxes for the snow-free period was observed for the RC sites post-management. However, the differences in mean daily N_2O fluxes for the snow-free period between post-management CT+WA and pre-management RC sites, as well as between post-management CT and pre-management RC sites were statistically non-significant.

In contrast, no significant relationships were found between stand age and mean daily N_2O-N fluxes for the snow-free period ($p > 0.05$; Fig. S3 C).

3.3. Environmental parameter impact on GHG fluxes

The models best describing the variation in the post-management mean daily GHG fluxes involved soil temperature at 10 cm depth, WTL, and management type as fixed effects in varying combinations (Table 3). Mean daily ER_{ff} could be best explained with soil temperature and soil WTL. Mean daily CH_4 fluxes were related to WTL and management type, while the variation of mean daily N_2O flux could be best explained with soil temperature, WTL, and management type. (Table 3).

3.4. The estimated total annual fluxes, and the impacts of forest management

Total annual GHG fluxes varied between the studied forest management types, generally aligning with the trends observed for the mean daily fluxes (Table 4). The lowest total annual ER_{ff} and CO_2eq were observed for the Control sites, followed by a partial cut, regeneration cut, commercial thinning, and combination of commercial thinning with wood ash fertilization. The regeneration cut produced total annual CH_4 emission of 11.55 CH_4-C, kg ha^{-1} yr^{-1}, while all other management types resulted in a soil CH_4 sink (Table 4). The strongest CH_4 sink was observed for commercial thinning, followed by a combination of commercial thinning with wood ash fertilization, Control, and partial cut. Total annual N_2O flux was the lowest in partial cut sites, followed by regeneration cut, commercial thinning, and Control. The highest total annual N_2O flux was observed for sites that underwent commercial

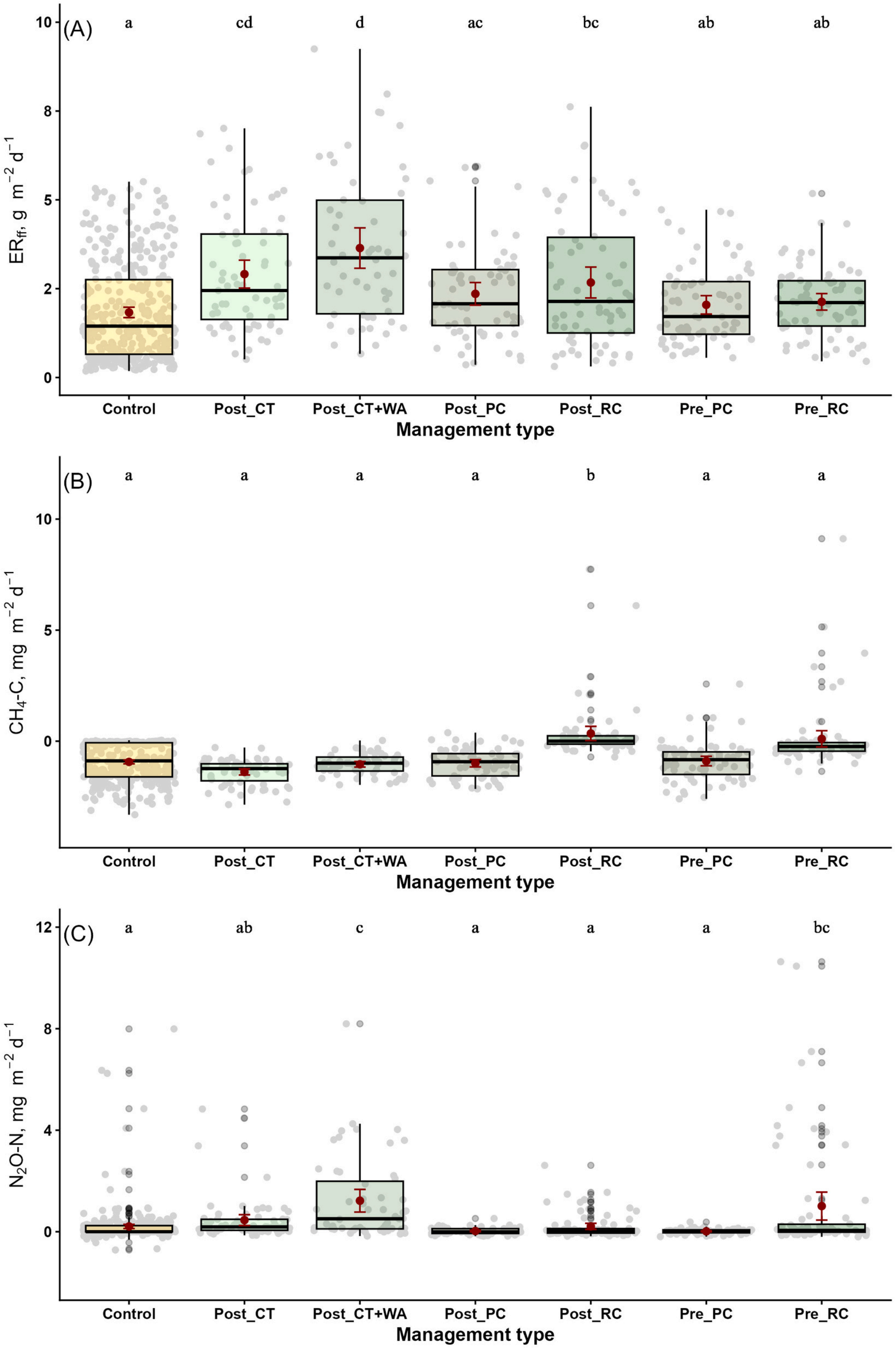

Fig. 4. Mean daily soil CO_2 (ER_{ff}, ecosystem respiration of forest floor), CH_4, and N_2O flux for the snow-free period for different management types prior and post management implication. Abbreviations: Control – forest that was not managed during the study period; Post CT – commercial thinning post-management; Post CT+WA – commercial thinning combined with wood ash fertilization post-management; PC – partial cut; RC – regeneration cut. In figures A, B, and C gray dots show single data points; red dots indicate *daily* mean values ± 95 % confidence intervals; the box plots represent *daily* median values and the 25 % and 75 % percentiles; the whiskers indicate the maximum and minimum values; and the black dots represent outliers. The letters in each figure indicate statistically significant differences in treatment means at the $p < 0.05$ level. Exact mean values are given in Table S3.

Table 3
Main effects of multiple environmental explanatory variables on post-management soil mean daily GHG fluxes (ER_{ff}, CH_4, N_2O) at site-level from the linear mixed effect models.

GHG flux	Explanatory variable	F value	Degrees of freedom	*p*-value	R^2
ER_{ff}	Soil temperature at 10 cm depth, °C	560.3	1	< 0.001	0.87
	Soil water-table level, cm	11.5	1	< 0.001	
CH_4 flux	Soil water-table level, cm	11.7	1	< 0.001	0.45
	Management type	7.4	4	< 0.001	
N_2O flux	Soil temperature at 10 cm depth, °C	5.0	1	< 0.05	0.32
	Soil water-table level, cm	6.2	1	< 0.05	
	Management type	1.9	4	< 0.05	

Table 4
Total annual soil ER_{ff}, CH_4-C, and N_2O-N fluxes, and the total annual soil flux as CO_2 equivalents, for unmanaged control sites and for the post-management period in study sites with different forest management types. Control –forest that was not managed during the study period and sites prior management; -CT – commercial thinning; -PC – partial cut; -RC – regeneration cut; -CT+WA – commercial thinning combined with wood ash fertilization. Note that the ER_{ff} is the total forest floor CO_2-C efflux including heterotrophic respiration and autotrophic respiration from ground vegetation and tree roots present in the monitoring points.

Management type	ER_{ff}, t ha^{-1} $year^{-1}$	CH_4-C, kg ha^{-1} $year^{-1}$	N_2O-N, kg ha^{-1} $year^{-1}$	CO_2eq, t ha^{-1} $year^{-1}$
Control	4.53 ± 0.82	-2.73 ± 1.46	1.74 ± 1.76	4.93 ± 1.05
CT	7.41	-4.92	1.65	7.73
CT+WA	8.95	-2.79	3.21	9.75
RC	6.82	11.55	0.67	7.31
PC	6.45	-2.59	0.09	6.41

thinning and wood ash fertilization.

4. Discussion

This is to the best of our knowledge the first study examining the impacts of forest management on GHG fluxes from drained organic forest soils in hemiboreal Europe, a region where drained peatland forests are common. Different forest management types examined in our study had influenced to a varying degree the site WTLs and soil temperatures and, consequently, the GHG fluxes. Forest floor respiration was also inevitably influenced by changes in ground vegetation biomass as well as tree root biomass that were not quantified in our study but have been documented elsewhere (Ryhti et al., 2021). Forest floor respiration in this study includes not only heterotrophic respiration and autotrophic root respiration but also an undetermined contribution from autotrophic respiration associated with the above-ground biomass of ground vegetation (forbs, graminoids, shrubs, mosses, and lichens), which is not included in many studies reporting soil respiration. The proportion of autotrophic respiration from both these sources can differ after different management types, and we have considered that in the interpretation of the results based on existing information on the extent of disturbance to the root and ground vegetation components by different management types. A clear distinction of inter-annual weather variation influence on GHG fluxes cannot be made due to experiment setup. However, monitoring years on average were warmer and drier than the 30-year mean, but similar to each other, thus inter-annual weather variation is not likely to affect the results to the extent that evaluation and conclusions of management impact is at risk (Fig. S1).

The ER_{ff} in our Control sites (Control) was within the reported flux range of ecosystem respiration of forest floor from organic and mineral forest soils in hemiboreal, boreal and temperate regions (Domisch et al., 2006; Ojanen et al., 2010; Korkiakoski et al., 2019; Butlers et al., 2025). The results confirmed the first hypothesis when forest management types were compared to Control sites: management increased the ER_{ff} from drained nutrient-rich organic soils. Mean annual forest floor respiration was 98 %, 64 %, 51 %, and 42 % higher than in Control sites in the CT+WA, CT, RC, and PC, respectively. However, there were no significant differences between the pre- and post-management ER_{ff} values within the RC and PC sites, likely due to the limited number of study sites and the resulting wide confidence intervals. The response of ER_{ff} to regeneration cut most likely indicates the effect of resulted disturbed ground vegetation and increased decomposition of roots and dead plant material on CO_2 flux. In addition, the increased soil temperature in RC sites should lead to higher CO_2 flux, while the elevated soil WTL would tend to reduce CO_2 flux from decomposition. Here, the overall impact of increased soil temperature and decomposition of roots and dead plant material on CO_2 flux was stronger than the flux reduction power of the elevated soil-water table level. The basic relationships between soil respiration, soil temperature, and WTL have been reported in earlier studies (e.g., Jungkunst et al., 2008; Köster et al., 2016). In the PC and CT sites, the disturbance to ground vegetation and roots is relatively minimal compared to the RC site, yet this disturbance contributes to resulted elevated fluxes likely due to the emissions from higher volume of decomposing roots. The elevated CO_2 fluxes following CT+WA management could similarly arise from increased decomposition of roots resulting from the thinning, combined with increased respiration of living roots and ground vegetation due to the fertilization (Zimmermann and Frey, 2002). Different forest floor responses to the applied management types indicates that some management types can be more effective in promoting climate change mitigation goals, i.e., via reduced soil disturbance and/or enhanced C storage in tree biomass. However, such differences should be verified with longer-term measurements.

When it comes to organic soils, there are only a limited number of studies available for comparison with our findings on management impacts. Studies of the boreal region in Finland have indicated lower rates of total ecosystem respiration and net ecosystem exchange (NEE) for partial cut (PC) compared to regeneration cut (RC) using the Eddy Covariance technique (Korkiakoski et al., 2023). Harvesting and thinning have been reported to have a slightly negative impact on soil C stocks, especially within the soil organic layer of forests growing on mineral soils. Moreover, thinning have been found to result in increased soil CO_2 fluxes shortly after management, particularly within the first two years (Mäkipää et al., 2023), as observed in CT sites also in our study. Specifically, light thinning has been reported to result in increased CO_2 respiration due to increase in root growth and autotrophic respiration, while heavy thinning has not shown increased flux rates, primarily due to countervailing factors such as elevated soil temperature and reduced litterfall and root biomass (Zhao et al., 2019; Mäkipää et al., 2023). Our finding that mean daily and annual CO_2 respiration was the highest in commercial thinning combined with wood ash fertilization (CT+WA) treatment aligns with other studies that report an increase in CO_2 flux after fertilization with wood ash (Mäkipää et al., 2023).

From a holistic perspective, forest management will thus result in higher ER_{ff} from peat soils. The evaluated impacts are, however, initial and possibly temporary. The Control sites in our study represented a range of forest stands, from the young to the old-growth stages, providing a comprehensive representation of mean GHG fluxes associated with control sites at various ages throughout the rotation period. Mean daily ER_{ff} for the snow-free period of control sites decreased with increasing site age (Fig. S3), likely driven by reductions in both root and microbial activity over time (Saiz et al., 2006). Moreover, the youngest Control sites, which have recently undergone regeneration show similar flux rates comparable to those of recently managed sites (RC and PC). This suggests that the management impacts on GHG fluxes persist and continue to exert influence during the early stages of stand development

(Fig. S3). However, it is important to note that in the majority of managed sites, the pre-management flux fall outside the confidence interval established for the Control sites indicating elevated microbial and root activity, possibly due to prior disturbances or microclimatic differences.

Accurate measurements of soil respiration are essential to determine the net ecosystem exchange and evaluate the ecosystem carbon balance, whether the ecosystem functions as a source or sink, and how management affects it. Earlier studies have reported that ecosystem carbon balance can change from source to sink in a relatively short time. Tree growth plays a main role in the ecosystem carbon balance of forest ecosystems, and after regeneration cut (RC) forest stands start to sequester more carbon than is lost from the ecosystem within ten years after harvesting (Grelle et al., 2023; Peichl et al., 2022). However, a partial cut site in Finland turned into a sink just three years after the cuttings (Korkiakoski et al., 2023). Further research is still required for describing and modeling the impacts of different management types on ecosystem and soil C balance in the longer term, over a full rotation or management cycle.

Our results of organic soil CH_4 uptake are consistent with a study in boreal forestry-drained peatlands (Ojanen et al., 2010), and are similar to the cutting impact study done in the boreal region, where the mean CH_4 flux of Control sites was from -0.65 CH_4-C mg m^{-2} d^{-1} to -0.84 CH_4-C mg m^{-2} d^{-1} (Korkiakoski et al., 2019; Korkiakoski et al., 2020). Most of the study sites continued to function as CH_4 sinks, even after implemented management, contrasting our second hypothesis, which is valid only for the RC sites. Only the regeneration cut (RC) site acted as a CH_4 source, due to the higher soil WTLs and reduced evapotranspiration resulting from the complete removal of tree cover (Leppä et al., 2020). A similar relationship between CH_4 flux and WTL has been reported in other studies, where relatively higher WTL results in increase in the CH_4 emissions (Matthews and Fung, 1987; Jungkunst et al., 2008; Ojanen et al., 2010; Korkiakoski et al., 2019). Higher soil WTL results in a thinner layer of oxic conditions within the soil, which can limit the oxidation of CH_4 produced in deeper anoxic peat layers (Sundh et al., 1995). Forest management may affect CH_4 flux by affecting the soil WTL. However, the results of this study suggest that, aside from these alterations to the WTL, the impact of management types on CH_4 flux is likely negligible. It has been suggested that the tree stand volume is a good scalar for CH_4 fluxes in peatland forests, as it generally closely correlates with the WTL (Minkkinen et al., 2007; Ojanen et al., 2019). Accordingly, the effect of logging on CH_4 flux is the largest immediately after harvesting, but gradually fluxes are likely to return to pre-harvest level (Saari et al., 2004; Peichl et al., 2022), which could also explain the significant differences between the pre- and post-management CH_4 flux of the RC sites. Therefore, possible measures to mitigate CH_4 emissions after the regeneration cut could be ditch system maintenance to lower the WTL. However, this may not be easy to achieve in sites where the soil is highly decomposed with high bulk density and low hydraulic conductivity (Gnatowski et al., 2010; Morris et al., 2015; Paat et al., 2022), as may be the case in old drainage areas (Laiho et al., 1999; Hyväluoma et al., 2020). Also, it is important to simultaneously consider the impacts of these management types on the other GHGs. The CT+WA treatment likely leads to increased plant growth and productivity due to the fertilization. This is expected to lower the soil WTL, and consequently, increase the CH_4 sink capacity of the soil following CT+WA (Ojanen et al., 2019).

More generally, studies from the boreal region have reported a range of CH_4 fluxes from drained organic soils, indicating both larger CH_4 sinks compared to our study and CH_4 sources, depending on soil WTL (Ojanen et al., 2010). Additionally, afforested sites on drained organic soils have shown smaller annual CH_4 uptake than what was observed in our study (Jauhiainen et al., 2023). The measured CH_4 flux includes ground vegetation CH_4 exchange, so it includes also the CH_4 released into the atmosphere by plants through various channels such as micropores in the stem, stomata, and intercellular spaces on leaves (Ge et al., 2024). Plant-mediated CH_4 fluxes from ground vegetation are the highest for aerenchymous sedges that benefit from wet conditions. Therefore, plant-mediated CH_4 fluxes may become notable after regeneration cuts especially if the rise in soil WTL is high enough to favor colonization by, e.g., cottongrass (Eriophorum vaginatum L.) (Hamberg et al., 2019; Kokkonen et al., 2019); otherwise they are likely to remain insignificant in forests.

The Control sites exhibited similar N_2O flux to the post-management levels of the PC and the RC treatments, therefore the third study hypothesis was not supported. The N_2O flux was the lowest in the partial cut (PC) site, where close to zero N_2O fluxes were observed throughout the measurement campaign. In our study, the CT and RC management types showed N_2O fluxes of similar magnitude to the Control, whereas the pre-management N_2O flux of the RC management type exhibited significantly higher values than both the Control and the post-management levels of the RC and PC treatments. Our RC sites demonstrated a significant decrease in N_2O flux. Clear-cutting in a boreal peatland forest resulted in a substantial increase in N_2O flux, rising from 1 to 228 ng N_2O m^{-2} s^{-1} (0.086–19.70 N_2O-N mg m^{-2} d^{-1}). In contrast, selection harvesting did not yield any significant impacts at the same study site. However, our findings contrast with previous reports of increased N_2O fluxes following regeneration cut (RC) (Korkiakoski et al., 2020). Higher pre-management N_2O flux compared to post-management levels in the RC sites likely indicates inter-annual variation in the fluxes, as observed in other studies (Korkiakoski et al., 2020), as well as the effects of clear-cutting. The introduction of younger vegetation following cutting likely enhanced nitrogen uptake and increased N demand compared to the 48-year-old pre-management stands, thereby reducing the pool of available soil N and contributing to the observed decline in N_2O emissions. Generally, soil N concentration, or availability correlates positively with the N_2O flux (Pärn et al., 2018; Minkkinen et al., 2020; Jauhiainen et al., 2023), while low P and Cu concentrations can limit N_2O production in peat even with sufficient N availability (Liimatainen et al., 2018). We observed the highest N_2O flux and soil N concentration for the CT+WA treatment sites. Typically, increased N_2O flux from peatlands following wood ash fertilization has not been observed (Ojanen et al., 2019); some studies even report a decrease in N_2O fluxes (Rütting et al., 2014). The higher N_2O fluxes after commercial thinning combined with wood ash fertilization in our study could thus likely be explained by the significantly higher soil N concentration in the CT+WA sites that may have facilitated higher N_2O fluxes as C:N ratio of the topsoil has an exponential relationship with N_2O emissions (Maljanen et al., 2006). Yet, the observed high mean N_2O fluxes after the CT+WA treatment in our study did not differ from the reported N_2O flux range of other studies evaluating the impact of wood ash on GHG fluxes (Maljanen et al., 2006; Champion et al., 2022). Moreover, annual N_2O flux in this study was similar to rates found in boreal nutrient-rich forestry-drained peatlands (Jauhiainen et al., 2023). Further studies should investigate the effects of wood ash treatment on GHG fluxes in organic soils, specifically analyzing variations across different soil N stocks and doses of wood ash. Overall, data on management impacts on the GHG fluxes from forests on drained organic soils are still too scarce for generalization and modeling (Jauhiainen et al., 2023), therefore, further studies should observe the long-term impact of different forest management types on soil GHG fluxes to assess GHG balance and recovery time after management-induced disturbances.

5. Conclusions

The study investigated the initial impact of forest management on greenhouse gas fluxes from drained organic soils in hemiboreal coniferous forests, and the objectives were met by highlighting the distinct responses of soil GHG fluxes to various forest management types and stand and environmental parameters. In unmanaged forests, forest floor CO_2 respiration was observed to decrease with increasing stand age. The results outline that various implemented forest management operations lead to different response in forest floor CO_2 respiration including soil and ground vegetation, mainly influenced by soil temperature and soil WTL. Commercial thinning combined with wood ash fertilization resulted in the highest forest floor CO_2 respiration. Throughout the observation period, most sites exhibited a CH_4 sink into the soil, except for those undergoing regeneration cut, where CH_4 emissions were observed due to an increase in WTLs. Furthermore, soil N_2O flux varied across different management types but likely responding to soil N concentration rather than the management, with partial cuts exhibiting the lowest values and commercial thinning combined with wood ash fertilization resulting in the highest N_2O flux due to higher soil N concentration. The N_2O flux was further influenced by soil temperature and WTL. Longer-term studies are essential to thoroughly evaluate the impacts of various forest management types on soil GHG fluxes, to assess their C balance and recovery periods needed after disturbances induced by management activities. Our study is one step towards increasing the understanding of how management alters the functions of the forest floors in these carbon-rich sites.

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CRediT authorship contribution statement

Valters Samariks: Writing – original draft, Visualization, Formal analysis, Data curation. **Raija Laiho:** Writing – review & editing, Supervision, Methodology, Conceptualization. **Aldis Butlers:** Writing – review & editing, Resources, Investigation, Data curation. **Andis Lazdiņš:** Supervision, Project administration, Methodology, Funding acquisition, Conceptualization. **Muhammad Kamil-Sardar:** Writing – review & editing. **Dovilė Čiuldienė:** Writing – review & editing. **Jyrki Jauhiainen:** Writing – review & editing, Supervision, Methodology. **Egidijus Vigricas:** Writing – review & editing. **Ieva Līcīte:** Supervision, Project administration, Methodology, Funding acquisition, Conceptualization. **Thomas Schindler:** Writing – review & editing, Methodology. **Kaido Soosaar:** Methodology, Conceptualization.

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.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.foreco.2025.123375.

Data availability

Data are available on Zenodo data repository (DOI: 10.5281/zenodo.13143011)

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