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The impact of large fires in boreal drained peatlands in western Finland: Ecohydrological drivers and carbon and nitrogen loss

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A R T I C L E I N F O

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A B S T R A C T

Climate change, characterized by more frequent drought periods along with anthropogenic activities, may increase the occurrence and severity of peatland wildfires in the boreal region. This study examines the impact of two large-scale peatland fires in 2020 and 2021 on carbon (C) and nitrogen (N) dynamics and losses in western Finland. The first site (Muhos, burned peatland area 217 ha) included both drained peatlands and pristine undrained mires, while the second (Susineva, burned peatland area 130 ha) was entirely drained for forestry. Our results reveal the significant impact of high-intensity fires on C and N dynamics and storage in drained peatlands, and in undrained mires with variable water regimes. In well-drained peatlands used for forestry, the average C and N losses during the fire were approximately 5.5 kg C m^{-2} and 123 g N m^{-2}. This estimated C loss exceeds the typical range reported for fire-events in undrained boreal mires. In contrast, the C loss in the undrained or poorly drained area fell within the range observed for undrained mires. The measured fire severity was influenced by drainage intensity and the types of vegetation communities. In undrained mires, the upper aerobic layer, with stable water regime, tends to burn only superficially or may even remain unburnt. However, in shallow mires with a variable water regime, where the surface peat dries extensively during drought periods, the C loss caused by fire was comparable to drained peatlands. Our findings underscore the importance of understanding the ecosystem services provided by peatlands, particularly considering management-related drivers such as drainage and fires. These factors can severely impact the peatland C balance and overall vulnerability, including reduced fire resilience.

1. Introduction

In the boreal region, peatlands play an important role in the global carbon (C) and nitrogen (N) budgets, containing 400–500 Pg of C as peat (Loisel et al., 2014), which is nearly equivalent to 50 % of the atmospheric CO_2 pool (Loisel et al., 2014; Le Quéré et al., 2018). In comparison, peatlands hold a smaller, yet ecologically important N pool estimated at about 10 Pg (Loisel et al., 2014). This amount is a small fraction relative to atmospheric N (3900 Pg) but constitutes approximately 10 % of the terrestrial N pool (Post et al., 1985; Limpens et al., 2006). Overall, N plays a crucial role in nutrient cycling and C sequestration.

Globally, the large C and N pool is vulnerable to climate and land-use change (Bowman et al., 2009; van der Werf et al., 2017). Fires are one of the most influential factors influencing soil organic matter dynamics (van der Werf et al., 2017). In peatlands, fires lead to substantial C losses and the release of significant amounts of greenhouse gases into the atmosphere (Kuhry, 1994; Pitkänen et al., 1999; Rodríguez Vásquez et al., 2021; Wilkinson et al., 2023). Fires emit the greenhouse gas CO_2, impact the flux of CH_4 and modify the climate by emitting precursors of aerosols and ozone, aerosols, and changing surface properties such as albedo (Randerson et al., 2006; Ward et al., 2012). Wildfire is also a major driver of N cycling increasing the soil emissions of nitric oxide (NO) and nitrous oxide (N_2O) (Gustine et al., 2022; Stephens and Homyak, 2023). This is particularly relevant in peatlands, where fire can intensify N losses in already vulnerable systems. N_2O is a powerful greenhouse gas, with a global warming potential 273 times greater than CO_2 over a 100-year period (IPCC, 2021). Drained peatlands, which are susceptible to

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fire, are global hotspots for N_2O emission, with emissions often exceeding those from most mineral soils (Tiemeyer et al., 2016; Pärn et al., 2018). Notably, Tiemeyer et al. (2016) estimate that about 72 % of N_2O emissions from organic soils originate from drained peatlands.

It has been estimated that wildfires annually burn approximately 1 % of the boreal forest and produce about 10 % of global CO_2 emissions, approximately 2 Gt C yr^{-1} (Zheng et al., 2021). Extreme wildfires, driven by rising temperatures are becoming more frequent and increasingly affecting the climate (Turetsky et al., 2011b; Huijnen et al., 2016; Liu et al., 2017; van der Werf et al., 2017; Rodríguez Vásquez et al., 2021; van der Velde et al., 2021). In boreal region, CO_2 emissions reached a new high in 2021, when the boreal fires produced a record-breaking level, 23 % of the total global CO_2 emissions into the atmosphere (Zheng et al., 2021). Peatland fires are a significant source of CO_2, which can be considered partially irreversible, as the C loss is not recoverable through vegetation regrowth or soil C accumulation within a human time scale (Zheng et al., 2021). This is particularly true for drained peatlands, where substantial greenhouse gas emissions from soils have been reported in areas converted for agriculture, forestry, and peat extraction (e.g., Kekkonen et al., 2019; Jauhiainen et al., 2023).

Peatlands are susceptible to fires globally, from tropical to arctic tundra regions (Turetsky et al., 2011b, 2015; Rodríguez Vásquez et al., 2021; Bourgeau-Chavez et al., 2020). In the boreal region, climatic factors regulated fires during the early and middle Holocene, while human activity took precedence during the late Holocene (Kuhry, 1994; Pitkänen et al., 2001; Olsson et al., 2010). In Fennoscandia, fire frequency increased during the postglacial thermal maximum (ca. 7000–5000 BP) and decreased in the mid-late Holocene due to a cooler, more humid climate. Fires surged again in the Late Holocene due to human activities like slash-and-burn agriculture (Clear et al., 2014). In the late 19th century, fires during dry summers in Finland could cover tens of thousands of hectares annually, mostly in dry upland forests (Lindberg et al., 2021). Nevertheless, most of the fires occurred in forests on upland mineral soil sites and peatland fires remained rare. According to the Finnish forest statistics (1911–1914), 5 % of the burnt area in southern Finland was in peatlands, while only 0.2 % of the burned area in northern Finland occurred in peatlands. The burned area has declined since then as forests became a valuable source of raw material, which led to increased efforts to prevent forest fires. From 1971 to 2019, Finland recorded only one fire event exceeding 200 ha, which occurred in 1997 (Lindberg et al., 2021).

Fire events in undrained boreal peatlands have been extensively studied in western Canada, particularly in relatively small peatlands with well-established tree (*Picea mariana*) and shrub layers (Kuhry, 1994; Robinson and Moore, 2000; Turetsky and Wieder, 2001; Turetsky et al., 2011b; Lukenbach et al., 2015; Hokanson et al., 2016; Wilkinson et al., 2023). In these studies, the reported average C loss is approximately 2.2 kg C m^{-2}, with values ranging from 0.5 to 6.5 kg C m^{-2}. Especially shallow peat deposits and peatland margins are found highly vulnerable to burning, and C losses up to 27 kg C m^{-2} have been reported, accounting for 50–90 % of total peatland C loss across various peat thicknesses (Lukenbach et al., 2015; Hokanson et al., 2016; Mayner et al., 2018; Wilkinson et. al., 2020). In Northern Europe, research has primarily focused on reconstructing fire histories through analyses of charcoal layers preserved in peat deposits with an average C loss of 2.5 kg C m^{-2} (Pitkänen et al., 1999), comparable to estimates from Canada.

Conversely, relatively few studies have focused on quantifying the total organic matter loss from drained boreal peatlands due to fire. Notable exceptions include studies by Turetsky et al. (2011a) and Wilkinson et al. (2018), who reported mean C losses ranging from 0.6 to 16.8 kg C m^{-2} from drained peatlands in western Canada. In temperate regions of Europe, relatively high C losses have also been recorded, for instance, 9.8 kg C m^{-2} from a partly drained, swampy peatland in Russia (Sirin et al., 2020), and 9.6 kg C m^{-2} from an afforested peatland experiencing smouldering fires in the UK (Davies et al., 2013). More recently, Wilkinson et al. (2023) modeled C emissions from northern peatlands by synthesizing published datasets, including data on degraded peatlands. Their findings indicate that peatland degradation significantly increases fire risk and C loss, with emissions reaching 10–25 kg C m^{-2} during wildfires. However, in Europe, there remains a significant research gap regarding the vulnerability of drained peatlands to burning during drought conditions, despite intensive peatland land use and rising fire activity driven by climate change.

Across Europe, approximately 48 % (174 000 km^2) of mires have been drained, mainly for agriculture and forestry (Tanneberger et al., 2021). In Finland, over half of the original 100 000 km^2 of mire area has been drained for forestry (Turunen and Valpola, 2020). Drainage can significantly alter the ecohydrological resilience of northern peatlands to fire, making them more vulnerable (Davies et al., 2016). The forestry-drained peatlands are particularly susceptible to fires during prolonged seasonal droughts due to water table drawdown (WTD), increased thickness of aerated surface layers, and the accumulation of flammable materials such as vegetation, wood, and litter biomass (e.g., Higuera et al., 2009; Vanha-Majamaa et al., 2021). This increased susceptibility may amplify fire intensity and its ecological impacts, making wildfire emissions from peatlands an underappreciated but potentially significant factor in C accounting (Wilkinson et al., 2023).

In Finland, climate change is predicted to increase wildfire frequency significantly by the end of this century. For example, Lehtonen et al. (2016) suggest that the number of large forest fires (> 10 ha) may double or even triple during the present century due to projected climate change. Similarly in Canada, the burned area is projected to increase by 74–118 % by the end of this century if wildfire is not limited by fuel availability (Flannigan et al., 2005). Although there is uncertainty of the projected forest-fire scenarios, there is a high probability for increased fire risk and some of the fires could develop into real conflagrations. For example, the extreme hot drought of 2018 that affected central and northern Europe led to the worst wildfire season in Sweden in over a century (Kelly et al., 2021). Across the whole country, 25000 ha of forest burned, which is ten times larger than the national annual average between 2000 and 2017 (SOU, 2019; Kelly et al., 2021). Similar increasing trends in the drought-enhanced fire activity during the late twentieth century have been observed in Canada and Alaska (Xiao and Zhuang, 2007).

In this study, we present data from the two largest wildfires that have occurred in Finland in recent years. Both fires were probably human-ignited after an exceptionally long warm period during 2020 and 2021. These fires attracted attention, because they were the largest wildfires (both about 250 ha) in 50 years but also included large, drained peatlands. Both fires were situated in the Ostrobothnia, where the landscape is flat with large peatlands, and where approximately 75 % of the peatland area has been drained (Toivonen et al., 2022).

The main aim of this study was to improve our understanding of the climatic impacts of widespread peatland fires on drained boreal peatlands. The specific objectives of the study were: (1) to compare the biomass losses due to burning in different environments (peatlands vs. dry upland forests, undrained mires vs. drained peatlands, seasonally dry vs. continuously moist undrained mires); and (2) to estimate the loss of C and N due to combustion resulting from a wildfire and human-induced activities in forestry-drained peatlands. We evaluate the impact of peatland fire on the C and N loss in forestry-drained peatlands with different drainage status. Our methodology relies on data derived from two peatland inventories conducted in 1999 and 2023, employing a systematic basin-based approach combined with resampling.

2. Material and methods

2.1. Site description of Muhos area

Muhos peatland (217 ha) is located in Muhos municipality in western Finland (Fig. 1). The characteristics of the study area are shown in Table 1. The Muhos-site includes dry, sandy upland forests and both

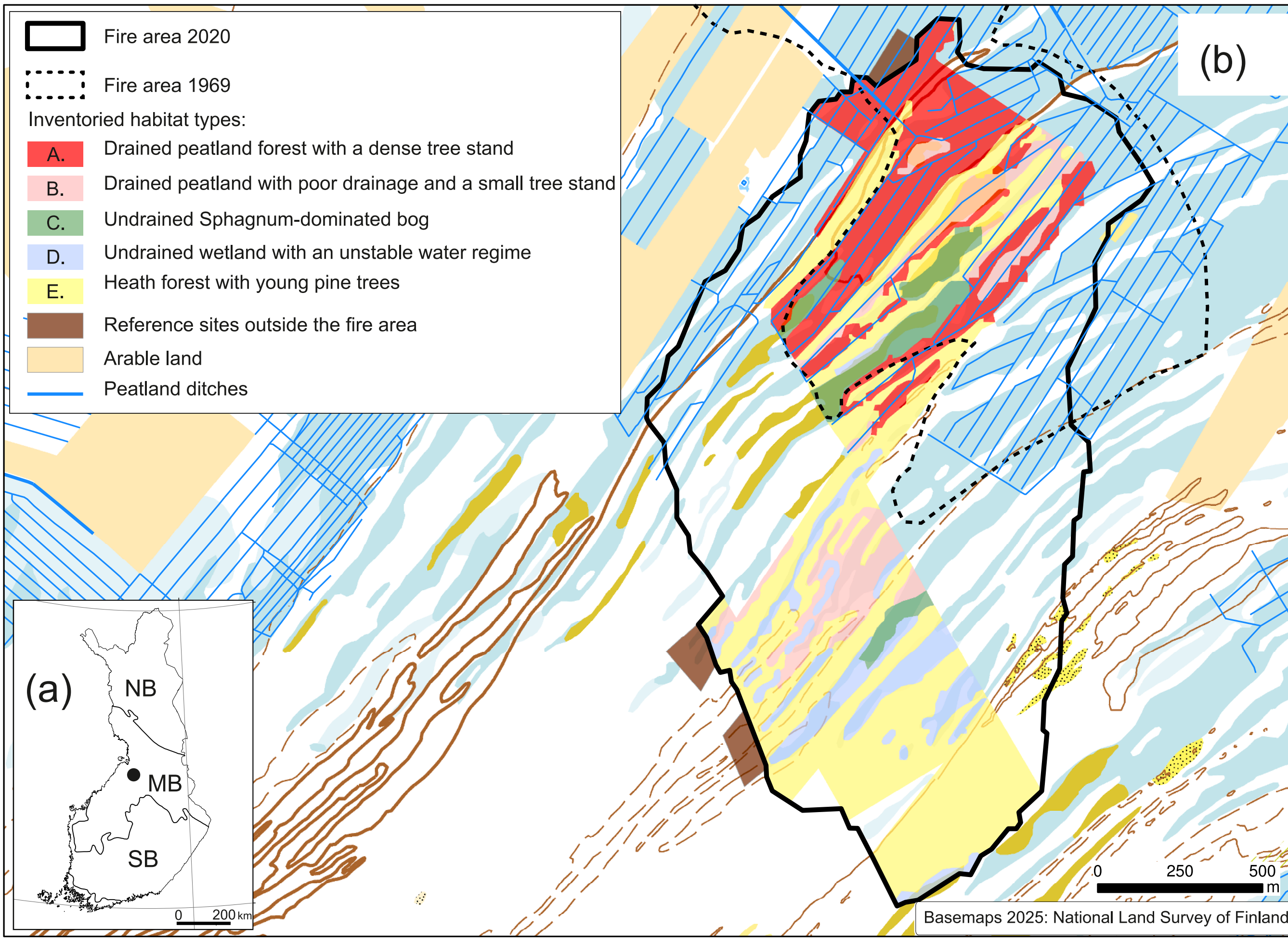

Fig. 1. (a). Muhos study site in western Finland. The boreal vegetation zone of Finland and its subzones are indicated (Hämet-Ahti, 1981): SB southern boreal, MB middle boreal, NB northern boreal. (b) The boundary of the 2020 fire area and the observed environment types are indicated with different colors.

Table 1
Characteristics of the study areas.

Peatland	North Lat.	East Long.	Altitude, m	Mean Temperature, °C June	July	Annual	Effective Temperature Sum dd	Mean rainfall June mm	July mm	Annual mm
Muhos	64°46,359′	25°56,774′	40–45	+13.3	+16.2	+3.2	1155	52	77	552
Susineva	64°00,272′	24°08,164′	70–74	+13.2	+16.0	+3.5	1122	59	74	542

For the climate at the study areas, climatological data of years 1990–2020 was used (Finnish Meteorological Institute 2023, https://www.ilmatieteenlaitos.fi/avoin-data). For effective temperature a + 5 °C threshold is used; dd is degree days.

drained and undrained peatlands. These peatlands are characterized by thin peat layers (30–100 cm) between sandy dunes and ridges (Laitinen, 2008), providing an opportunity to compare the effect of fire across different ecosystem types. The bedrock consists mainly of silicate-siltstone, which is covered by thick layers of sand, till and other loose sediments (Superficial deposits and bedrock of Finland, https://hakku.gtk.fi/). In the paludified depressions, water percolates through the damming beach ridges, creating unstable aro-wetland ecosystems (Laitinen et al., 2008) with little or no peat layer (0–30 cm). However, areas of the basins with impermeable bottoms have developed into weakly minerotrophic *Sphagnum* peatlands or *Carex* fens with thicker peat layers (> 30 cm).

The upland forest sites on the sand bars are primarily dry and very nutrient-poor *Cladonia*-type sites, whereas the forestry drained peatlands with dense tree stand are more nutrient-rich, featuring *Vaccinium vitis-idaea* and *Vaccinium myrtillus* types (for classification of forestry-drained peatlands, see e.g., Vasander and Laine, 2008). The forested sites were mainly occupied by even-aged Scots pine (*Pinus sylvestris* L.) stands. The depressions began to paludify largely just after they rose above the sea level 4000–4500 years ago (Rehell and Virtanen, 2015). Most of the drainage occurred at the Muhos-site in the 1970 s. Some poor, periodically water-covered undrained basins were ditched later, only a few years before the 2020 fire. The ditch spacing varies but is generally between 40 and 50 m in well-drained forestry-drained areas.

The Muhos wildfire in 2020 burned 251 ha, including 217 ha of peatlands. It ignited on June 29 in dry conditions, spreading rapidly as a crown fire. The fire occurred after a dry spring and an exceptionally warm June, with prolonged heatwaves. In Muhos area, the average temperature of June, 16.9 °C was 3.6 °C above the 1991–2020 average, making it the warmest June on record (Table 1). Also, the maximum day temperature in June was regularly high with a mean of 22.3 °C and a maximum of 29.4 °C (Finnish Meteorological Institute, 2023). Moreover, the total rainfall of June was low, 37 mm, approximately 29 % less than the average (Finnish Meteorological Institute, 2023). The fire-extinguishing efforts lasted for one week and were aided by rainfall.

In Muhos-site, nearly the same area burned about 50 years ago. In 1969, the fire area was altogether large, about 2000 ha. The burnt area was mainly upland forests but also included drained minerotrophic fens mostly treeless at the time (Fig. 1b). The undrained *Sphagnum*-covered mires remained unburnt until they were drained for forestry and deeply burned in 2020. The Muhos peatland was investigated by the Geological Survey of Finland (GTK) in 1981 using a few research points, but no laboratory analyses. The peatland was described as having humified, mainly *Carex*-dominated peat deposits (Häikiö et al., 1983).

2.2. Inventories in the Muhos-site

In summer 2022, Metsähallitus made an inventory covering 130 ha within the fire-affected area. To minimize measurement inaccuracies, a similar study was also carried out on a 4-ha reference area outside the fire zone (Fig. 1). The fire area was subdivided into 170 forest stand units, which were made as homogenous as possible in terms of tree structure and surface vegetation, based on field observations and aided by aerial photographs. Measurements were made at all sites to determine the common tree stand characteristics such as stand basal area (m^2 ha^{-1}, the total stand volume (m^3 ha^{-1}), mean tree height (m) and mean tree diameter (cm) at breast height (1.3 m), tree mortality and peat or humus layer thickness. The depth of burn was measured at three random spots within each of the 170 units ($n = 510$). The depth of peat consumption by fire was estimated following the method described by Kasischke et al. (2008), by measuring the vertical distance from the post-fire soil surface to the root collar of pine stumps. This approach assumes that the root collar marks the pre-fire ground level, and thus the measured distance represents the depth of peat burned during the fire.

The C and N contents of the burnt surface peat were estimated using average values for surface peat from various peatlands across Finland (Turunen et al., 2024; Table 2). Mean C and N concentrations of 52.3 % and 2 % were applied for the peat samples (Päivänen, 2007; Turunen and Valpola, 2020).

The density of the dry organic wood material in the tree stands was estimated to be approximately 460 kg m^{-3}. According to Laine and Vasander (1996) and Päivänen (2007), the above-ground biomass of field and bottom layers of the surface vegetation consisted mainly of dwarfs and herbs was estimated to be around 0.2 kg m^{-2} on undrained mires and around 0.2 kg m^{-2} on drained peatland forests. Field observations indicated that the fire completely burned the vegetation on drained mires, but only in patches on undrained mires. Therefore, it was assumed that the fire burned approximately half of the field layer vegetation on undrained mires. The C content of dry organic humus and wood material was estimated to be 50 % (Mäkinen et al., 2006), while the N content was estimated to be about 2 % of the dry weight in humus, 1 % in field layer vegetation (small dwarf-shrubs), and about 0.5 % in trees. In forests on mineral soil sites in uplands, the biomass of field layer vegetation was estimated to be about 0.1 kg m^{-2}, and the humus about 2.5 kg m^{-2} (Lindberg et al., 2011). In upland forests, it was assumed that the fire burned 73 % of the humus layer (Wilkinson et al., 2020).

2.3. Site description of Susineva peatland site

The Susineva peatland (180 ha) is located in Kalajoki municipality in western Finland (Fig. 2). The characteristics of the study area are shown in Table 1.The Susineva peatland was largely well-drained for forestry purposes mainly at the shift of the 1960s and 1970s with a 40-m ditch spacing, which is typical for Finnish drained peatlands. Most of the drained sites in Susineva can be currently classified to nutrient-poor dwarf shrub peatlands with Scots pine (Vasander and Laine, 2008). Before drainage, the sites were treeless or sparsely forested, *Sphagnum* sp. dominated ombro-oligotrohic pine bogs. In the western part of the peatland, there remains a small, poorly-drained area (13 ha) with low-productive forests (about 15 m^3 ha^{-1}), isolated from its catchment by surrounding ditches (referred to as area 1, Fig. 2b-c). This has been the poorest and wettest part of the mire complex, occupied by mainly open treeless mires. The mineral subsoil in the central part of the peatland consists of silt, while the marginal areas are composed of till. The adjacent upland soils are characterized by boulder till. The bedrock of the area consists of granodiorite (Superficial deposits and bedrock of Finland, https://hakku.gtk.fi/). Before the fire, the peatland area consisted of evenly aged (25–30 years old) Scots pine stands, which had regenerated naturally after drainage without any harvesting or replanting. In the mineral soil area, the thickness of the humus layer was around 10 cm.

The Susineva peatland is also part of the largest wind power park in Finland. Construction of the park, called Mutkalampi, started in 2021 including an extensive road network around the peatland. During the road construction work, a fire started in the afternoon of July 26th, 2021 (Saarela, 2022). The weather conditions (June-July) were favorable for the fire including drought conditions and gusty wind. In Central Ostrobothnia, western Finland, the mean July temperature was 18.3 °C, well over the average, and the total rainfall was relatively low 56 mm (Table 1). Moreover, the maximum day temperature in July was regularly high with a mean of 24.0 °C and a maximum of 29.4 °C (Finnish Meteorological Institute, 2023). Initially, the fire burned 45 ha of upland forest before being extinguished. However, on July 28th, unnoticed smouldering reignited due to gusty winds, rapidly spreading west as a high-intensity crown fire towards Susineva peatland, ultimately being extinguished by rain on July 31st. The total extent of the fire was approximately 227 ha including 130 ha in the Susineva peatland. Overall, the suppression efforts lasted about two weeks, involving approximately 1500 personnel and incurring costs of around 2.3 million euros (Saarela, 2022), making it an exceptionally costly operation. In 2022, a 90-ha area of the Susineva was designated as a temporary protected site for a period of 20 years (Fig. 2c).

2.4. Peatland inventories before and after fire in the Susineva-site

Comprehensive information regarding this drained peatland both pre- and post-fire is accessible through the peatland inventory conducted by GTK. The first peatland research of Susineva was completed in 1999 by GTK using a transect-method (Fig. 2b). Each transect consists of research points, where a detailed analysis was done including peatland

Table 2
Surface peat dry bulk density (dBD) characteristics (mean ± SE) of the study areas A-D in Muhos site. The dBD of the burnt surface peat was estimated using the mean values from Turunen et al. (2024) for different types of peatlands.

Ecosystem type (see Fig. 1.)	dBD kg m^{-3}	n	Corresponding surface peat type in Turunen et al. (2024)
A. Drained peatland with dense tree stand	98.1 (2.3)	196	Drained *Carex*-peat
B. Drained peatland with sparse tree stand	80.8 (2.5)	211	Drained *Sphagnum*-peat
C. Undrained *Sphagnum* dominated bog	52.7 (3.1)	83	Undrained *Sphagnum*-peat
D. Undrained wetland with unstable water regime	61.6 (6.3)	15	Undrained *Carex*-peat

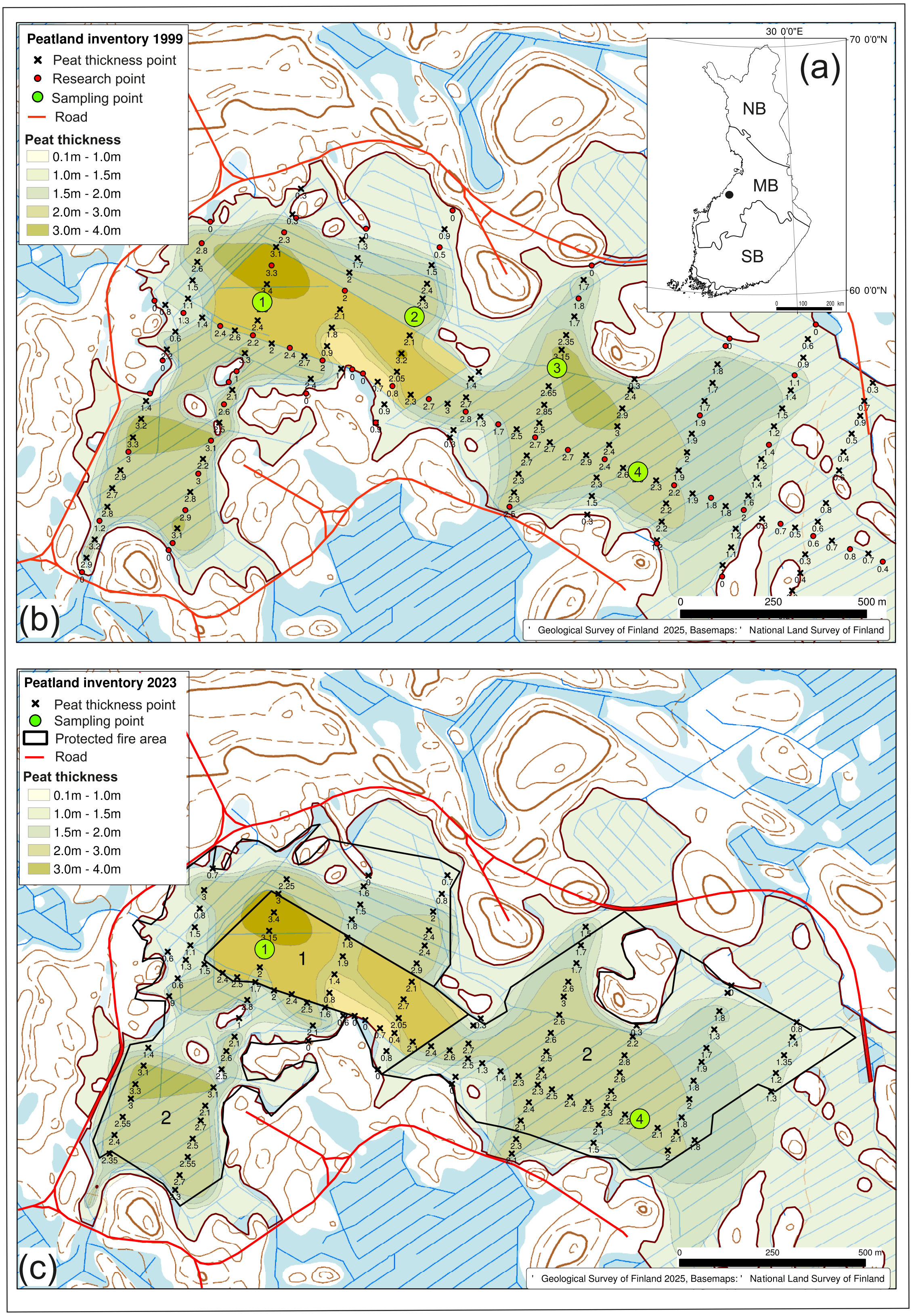

Fig. 2. (a) Susineva peatland in western Finland. The boreal vegetation zone of Finland and its subzones are indicated (Hämet-Ahti, 1981): SB southern boreal, MB middle boreal, NB northern boreal. (b) Peatland inventory 1999 and (c) in 2023 with study areas 1 and 2 within the protected peatland area, transects, peatland thickness areas and sampling sites indicated.

type, drainage status, vegetation, peat types, peat thickness, existing gyttja layers and subsoil type under the peat deposits. In the field, the peat type was determined based on the botanical composition (Lappalainen et al., 1984), where the composition of the sample is determined in 6-grade scale as *Sphagnum, Carex or* Bryales peat according to the most abundant plant residues. The degree of decomposition was estimated with von Post's (1922) 10-grade scale, ($H_{1\text{-}10}$). Within the fire area, a total of 50 research points were established, with peat thickness additionally measured at 75 thickness points along the transects (Fig. 2b). In 2023, a re-inventory was done in the peatland fire area using the corresponding points as in 1999 along the original transects (Fig. 2c). At every point, the residual peat thickness was measured.

In 1999, the continuous volumetric peat profiles were collected from four (4) sampling sites with a piston corer (80 × 200 mm, Fig. 2b) for dry bulk density, water and ash content analysis. In 2023, the samples were collected from two (2) corresponding sites contiguously from the surface down to the bottom of the peat layer with a box sampler (70 × 70 × 1000 mm) and with a Russian pattern side-cutting peat sampler (50 × 500 mm, Fig. 2c) for comparison. Great care was taken over the volumetric accuracy of the samples (Pitkänen et al., 2011). All samplers used were made of stainless steel. The 20-cm peat samples (0–30 cm surface peat in 1999) were collected in plastic bags and stored in dark and cool conditions (+4 °C) before analysis.

In autumn 2022, Metsähallitus made an inventory of the protected area using the same methods as in the Muhos-site. Observations indicated that the central and southern parts were completely burned, with all trees dead and the ground layer appearing still sparse, featuring species such as *Marchantia polymorpha* and *Polytrichum strictum* on the burned surfaces. In the small, poorly-drained area 1 and in some parts north of it (Fig. 2c), there were more superficially burned patches with some remaining peatland vegetation features, such as *Sphagnum fuscum* hummocks that were only burned at the edges. On the mineral soil islands within the Susineva site, the humus layer was largely burned away, exposing the underlying soil along with large rocks and boulders.

2.5. Laboratory analyses

The peat samples collected in 1999 (n = 52, Fig. 2b) were analyzed by the GTK Laboratory in Kuopio in 1999, while those collected in 2023 (n = 24, Fig. 2c) by the Eurofins Labtium Laboratory in Jyväskylä, Finland. The peat samples were dried to a constant mass at 105 °C and weighed. The water content was determined using the gravimetric method by weighing the wet soil sample, drying it in an oven, reweighing and calculating the mass of water lost as a percentage of the mass of the dried soil. The dry bulk density (kg m^{-3}) of the samples was calculated by dividing the dry peat mass by the fresh volume (m^{-3}). The peat samples were milled, and the ash content was determined as loss-on-ignition (LOI) at 815 °C. For the 2023 samples, the C and N concentrations were analyzed with LECO 628 CHN analyzer. Samples were heated, oxidizing C to CO_2 and N to N_2. The amounts of C and N were detected with an infra-red detector (NDIR) and thermal conductivity cell (TC), respectively. The element density (kg m^{-3}) was calculated by multiplying the dry bulk density by the corresponding C or N element concentration. In 2024, the surface peat samples (0–30 cm) from four 1999 profiles were retrieved from the GTK peat archive, where they had been stored in sealed plastic bags in a dry, light-protected environment, and analyzed for C and N using the same methods as the 2023 samples. For the rest of the samples collected in 1999, the average C and N concentrations measured in 2023 were applied.

2.6. Peat density calculations

For peat profiles collected in 1999, we estimated the dry bulk density values for the top 0–30 cm samples, for which only the water content (%), ash percentage of dry matter (%) and the von Post humification index ($H_{1\text{-}10}$, von Post, 1922) were reported. To make the estimation, we developed a regression model (1) using 473 surface peat samples (0–40 cm) from forestry-drained peatland sites in the GTK database, following a methodology similar to that of Simola et al. (2012). We also made alternative C loss calculations using the average surface peat (0–40 cm) dry bulk density values reported by Turunen et al. (2024) for undrained and drained ombro–oligotrophic forested bogs, which are comparable peatland types found in Susineva.

$$\mathrm{dBD} = 721.4 - (7.2\,\mathrm{x}\,\mathrm{w\%}) + (3.9\,\mathrm{x}\,\mathrm{H}_{1-10}),\ \text{adjusted}\ \mathrm{R}^2 = 0.64 \quad (1)$$

where dBD = dry bulk density (kg m^{-3}), w% = water content of the peat sample (% of fresh mass) and $H_{1\text{-}10}$ = von Post humification index (von Post, 1922). The 95 % confidence intervals were taken as error estimates for the reconstructed dBD.

2.7. Estimation of the carbon and nitrogen mass changes

For the estimation of peat mass loss caused by fire, we used two different approaches. First, starting with a simple approach, the average thickness change of the peat deposits between two inventories (1999, 2023), the surface dry bulk density values of 1999 inventory before the fire (sites 1–4, Fig. 2b) and the corresponding surface peat C and N concentrations were applied for areas 1 (sites 1–2, Fig. 2b) and 2 (sites 3–4, Fig. 2b) separately.

Second, a comparative approach was done based on the total peat volume obtained from the peat thickness measurement in 1999 and 2023. The total peat mass in 1999 and 2023 was evaluated by interpolating the peat volume in different peat thickness zones in two phases: First, the peat thickness zones with 0.5 – 1.0 m depth classes were drawn based on the systematic peat thickness inventory (Fig. 2b-c). Second, the peat volume was modelled by implementing a simple model developed in GTK (Hänninen et al., 1983). The volume of peat, degree of decomposition, as well as the proportions of peat types and peat-forming factors have been calculated using the so-called zonal calculation method. In this method, each area between two adjacent contour lines or between a contour line and the edge of the peatland forms its own thickness zone. The total peat volume (V) was calculated by equations 2–4. First, the peat volume was calculated separately for each thickness zone (V_z) as in the following:

$$V_z = \frac{\sum_{i=1}^{n_z} L^t + \sum_{i=1}^{k_z} L^s}{n_z + k_z} A_z \quad (2)$$

where L^t is the length of the research point, L^s is the length of the thickness point, n_z is the number of research points in zone z, k_z is the number of depth points in zone z, and A_z is the surface area of zone z. Second, by combining the equal thickness zones we will get the total peat volume of each thickness zone (V_h):

$$V_h = \sum_{i=h}^{n} V_z \quad (3)$$

where n is the number of thickness zones. Finally, the total peat volume of the whole peatland area is obtained by summing-up all thickness zones in the peatland area using equation (4):

$$V = \sum_{i=1}^{n} V_h \quad (4)$$

where *n* is the number of different thickness zones. The degrees of decomposition as well as the quantities of peat types and peat-forming factors are calculated, weighted by the peat quantities.

To determine the C and N mass of the areas 1 and 2 in 1999 and 2023, we used the mean dry bulk density values calculated from two corresponding sampling sites (sites 1 and 4, Fig. 2b-c). The mean C and N concentrations measured from the samples collected in 2023 were also applied for peat profiles collected in 1999. The Mann-Whitney *U* test was used to test the equality of physical properties between 1999 and 2023.

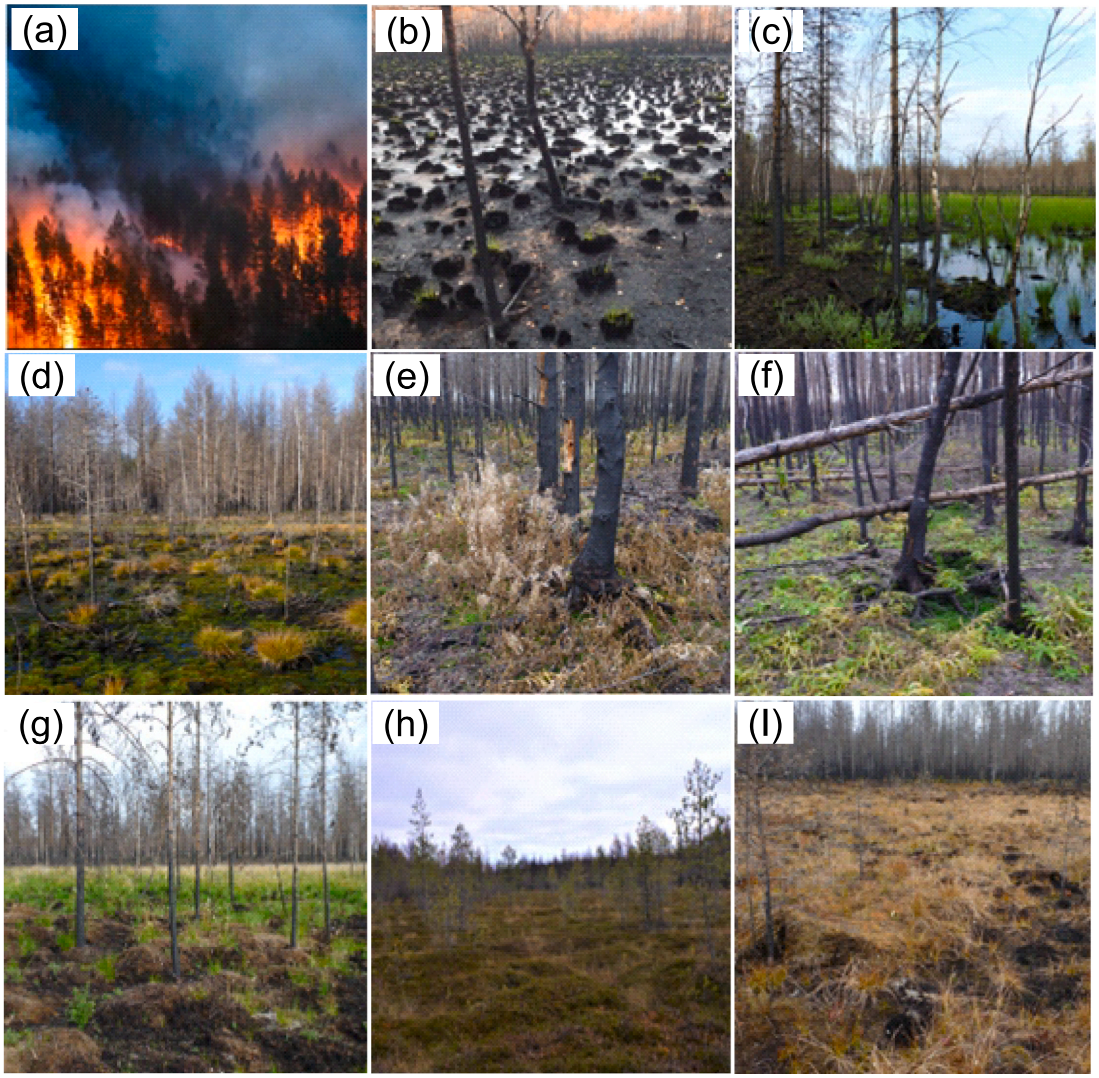

Fig. 3. Photos from the Muhos fire area. (a) Uncontrollable crown fire (photo the Utti Jaeger Regiment), (b) Undrained mire area in the center of the burnt area two weeks after the fire. The mire has lost its thin peat layer (photo E. Melantie), (c) Mire area in the center of the burnt area one year after the fire, (d) Drained peatland forest and the dried-out pristine low-sedge fen in the burnt area one year after the fire, (e-f) Drained peatland forest one year after the fire, where the surface peat had burned deeply, especially near the trees, but the wood material remained unburned, (g) Undrained *Eriophorum vaginatum* pine bog, where the lawn and carpet levels were severely burned, but on the hummock-level *Sphagnum* mosses were alive one year after the fire, (h) In the center of the fire area, an undrained mire pattern, dominated by *Sphagnum fuscum,* remained completely unburnt, (i) On the undrained lawn-level treeless mire, *Sphagnum* mosses were burned near the drained peatland forest but were largely unburnt already about 50 m from the nearest ditch. Photos S. Rehell unless otherwise indicated.

Table 3
Field inventory results of the Muhos area. See Fig. 1 for additional context.

Environment type	Fire area %	Mean burn depth (range) cm	Resid peat depth cm	C loss of peat / humus kg m^{-2}	C loss of field & ground floor veg. kg m^{-2}	N loss of peat / humus g m^{-2}	N loss of veg. g m^{-2}	Tree stand m^3 ha^{-1}	Dead trees in fire %	C loss of trees kg m^{-2}
A. Drained peatland with dense tree stand	39	10 (5–30)	11–80	5.3	0.2	210	8	71	98	0.2
B. Drained peatland with sparse tree stand	11	7 (0–19)	1–90	3.1	0.2	76	8	9	100	0.04
C. Undrained *Sphagnum* dominated peatland	3	1 (0–10)	60–110	0.3	0.1	10	4	1	90	0.002
D. Undrained wetland with unstable water regime	10	9 (5–25)	0–30	3.6	0.1	100	4	34	94	0.08
E. Upland forest	37	2 (0–5)	–	0.8	0.2	2	6	67	92	0.2

3. Results

3.1. The impact of fire in the Muhos-site

The 2020 fire in the Muhos-site can be categorized as a high-intensity fire, which progressed as an uncontrollable crown fire (Fig. 3a). The depth of burn was notably significant on the drained peatlands but also on some undrained mires with variable hydrological conditions (Fig. 3b-i). On these thin-peated sites with naturally variable water regime, large water bodies covered the former peatland areas after the fire. In drained peatland forests, the peat was unevenly consumed. Practically all trees died in both drained peatlands and upland forests. Although the trees died, only their needles, small twigs, and dead parts of the trees burned, accounting for about 0.6 kg m^{-2} or 10 % of the tree volume; the rest of the wood material remained nearly unburned (Fig. 3h). However, even in the center of the fire area, undrained mires with a stable water regime burned only superficially. One ombrotrophic *Sphagnum fuscum* bog pattern, about 3.2 ha, was left nearly totally unburned in the Muhos-site. Reference patterns inventoried outside the 2020 fire area had an average tree stand volume of 77 m^3 ha^{-1}, with an amount of dead wood of 1.4 m^3 ha^{-1}.

The field inventory results, and the C and N loss estimates for the Muhos-site are presented in Table 3 and Fig. 4. Across the inventoried 130 ha area, the average area-weighted total C loss was 2.5 kg C m^{-2} (including peat, humus, field and ground vegetation, and trees), distributed as follows: A. Drained peatland forests with a dense tree stand contributed 5.5 kg C m^{-2} (49 %), B. Drained peatlands with poor tree stand 3.3 kg C m^{-2} (19 %), C. Undrained *Sphagnum*-dominated peatlands 0.4 kg C m^{-2} (1 %), D. Undrained wetlands with unstable water regimes 3.7 kg C m^{-2} (14 %), and E. Upland forests 1.2 kg C m^{-2} (18 %) (Fig. 4). Across the entire burned area, the majority of total C loss originated from drained peatlands (68 %), while undrained peatlands and upland forests accounted for 15 % and 18 %, respectively.

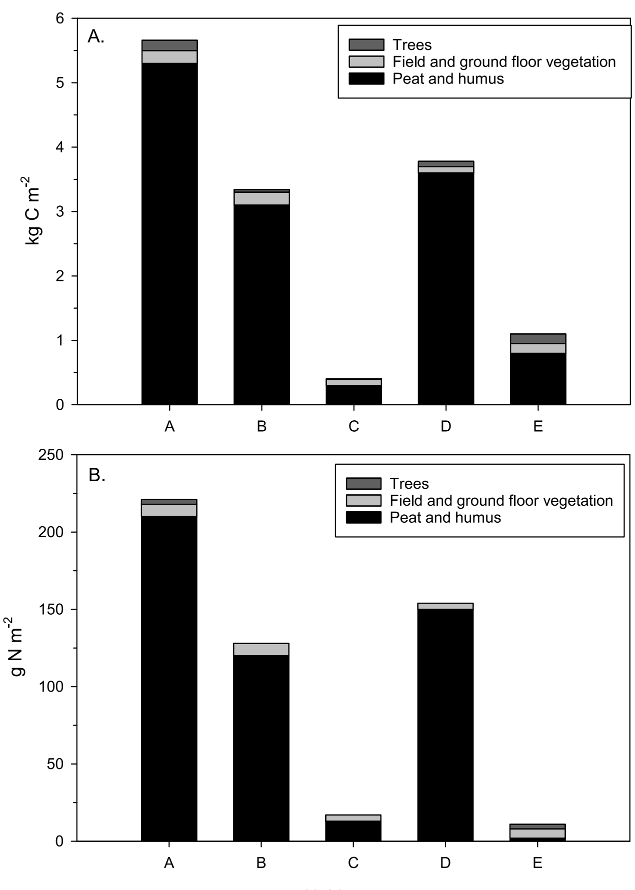

Fig. 4. Carbon (g C m^{-2}) and nitrogen loss (g N m^{-2}) estimates for the Muhos-site (including trees, field and ground vegetation, peat and humus). A. Drained peatland forests with a dense tree stand, B. Drained peatlands with poor drainage, C. Undrained *Sphagnum*-dominated peatlands, D. Undrained wetlands with unstable water regimes, and E. Upland forests.

Overall, approximately 88 % of C emissions were derived from peat and humus layers, 7 % from ground vegetation, and 5 % from trees. The highest average peat C loss, 5.3 kg C m^{-2}, occurred in a well-drained peatland forest with a 40–50 m ditch spacing (referred to as area A in Fig. 1). Notably, 94 % of the C loss from undrained soils originated from wetlands with unstable water regimes.

N loss exhibited a similar pattern to C loss (Table 3). The average area-weighted total N loss from the study area was 92 g N m^{-2}, while the soil-specific N loss averaged 85 g N m^{-2}. Drained peatlands contributed 180 g N m^{-2}, undrained peatlands 99 g N m^{-2}, and upland forests 10 g N m^{-2}. Consistent with C loss, approximately 94 % of the N loss from undrained soils was attributed to wetlands with unstable water regimes.

3.2. The impact of fire in Susineva

3.2.1. Peat characteristics

In the Susineva-site, the peat deposits were moderately decomposed with an average von Post value of H_5 ($H_{1\text{-}10}$, von Post, 1922). In area 1, the original surface peat (0–50 cm) has been weakly decomposed ($H_{1\text{-}3}$) *Sphagnum* peat, whereas the deeper peat deposits are *Carex* dominated with high N concentrations (> 2 %) and a low C:N ratio around 25 (Fig. 5d-e). In area 2, the peat deposits have a relatively low N concentrations and high C:N ratio throughout the peat deposits, which are *Sphagnum* dominated (Fig. 5i-j).

For peatland inventory 1999, the estimated surface (0–30 cm) dry bulk density values (mean ± SE) from two western sampling sites for area 1 and from two eastern sampling sites for area 2 (Fig. 2b) were 32.3 ± 1.1 and 77.5 ± 3.4 kg m^{-3}, respectively. The corresponding dry bulk density values used for alternative calculations were 44.9 ± 5.9 kg m^{-3} (n = 26) for undrained bogs and 71.4 ± 3.0 kg m^{-3} (n = 107) for drained bogs, as reported by Turunen et al. (2024). The analyzed C and N concentrations were 48.6 ± 1.0 % and 0.94 ± 0.10 % for area 1, and 50.4 ± 1.8 % and 1.30 ± 0.38 % for area 2.

The average changes in peat physical properties across entire peat profiles from 1999 to 2023 were relatively minimal (Fig. 5). Statistically significant differences (Mann-Whitney *U* test, $p < 0.05$) were not found in physical properties between 1999 and 2023. However, there was a slight decrease in water content and increase in the dry bulk density and ash content in 2023 (Fig. 5). Also, in area 1, the dry bulk density profile of the top 100 cm showed signs of subsidence with increased dry bulk density values (Fig. 5b). The C concentrations of both profiles were also relatively uniform with an average 54.5 ± 0.5 %.

Based on the individual peat thickness measurements (n = 108), the average thickness of the peat deposits in 2023 inventory had decreased by 18 cm compared to the 1999 inventory (Fig. 6). Depth of burn ranged from 0 to 60 cm within the burned site. In total, across nine study sites, we observed an increase in peat thickness ranging from 10 to 30 cm. Weighted by the peat thickness areas, the average decrease of the research area 1 and 2 were 12 and 15 cm, respectively (Table 4). A total of 17 thickness measurements, primarily near the western mire margins, were excluded from both the 1999 and 2023 inventories due to measurement uncertainties, as the subsoil at these study points includes large rocks and boulders. In area 2, there was more variation in the fire severity within the whole area. Notably, the relatively shallow peat deposits near the peatland margins and close the mineral soil islands within the peatland had burned most intensively (Fig. 7a-g).

3.2.2. Carbon and nitrogen losses

Based on the surface peat mass loss estimation, the area weighted value (mean ± SE) for C and N loss for the entire peatland basin of the

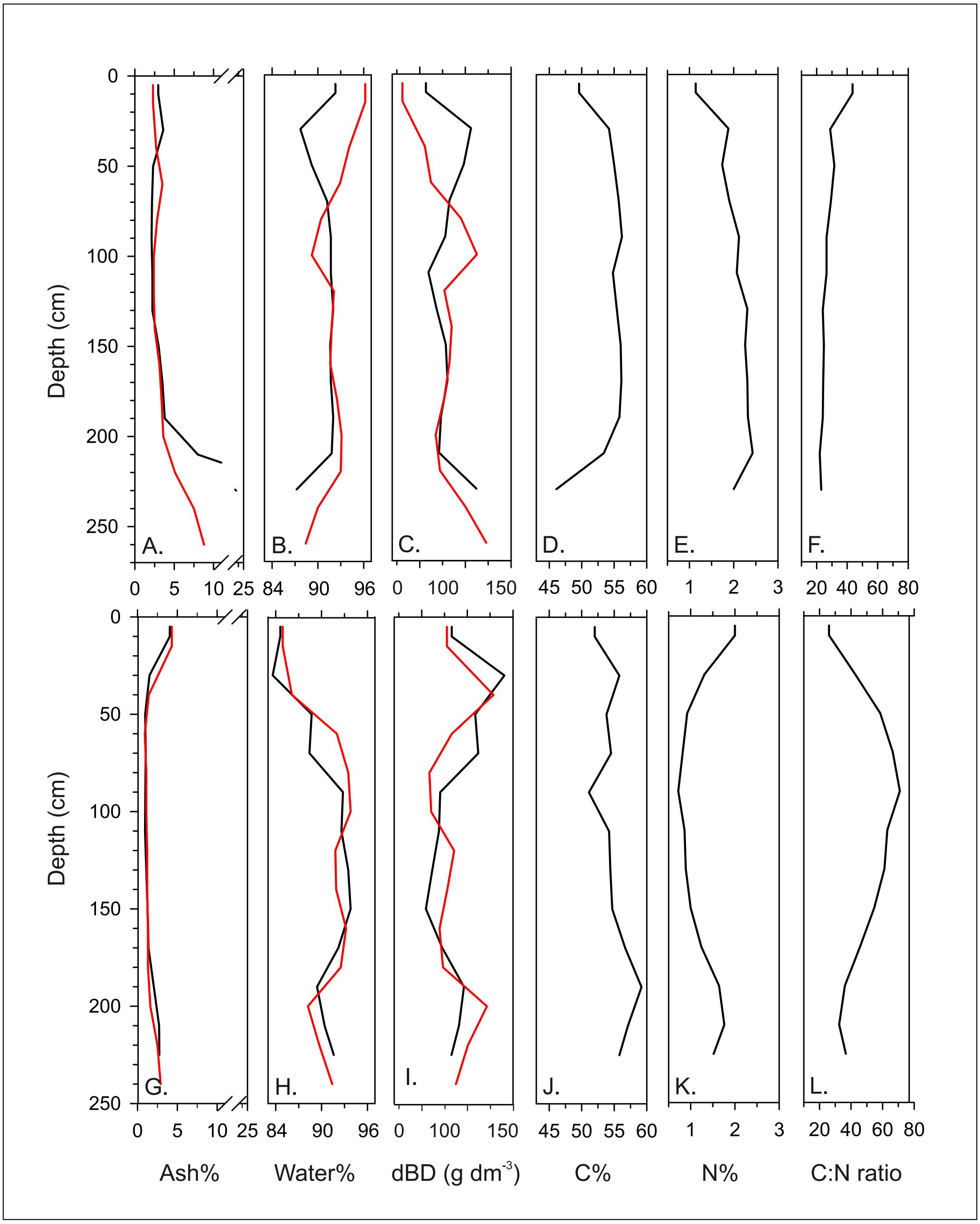

Fig. 5. Ash content (%), water content (%), dry bulk density (g cm^{-3}), carbon and nitrogen concentrations (%) and C:N ratio of peat in study area 1 (a-f) and area 2 (g-l) in Susineva peatland site. Red curves = inventoried in 1999 (n = 25), black curves = inventoried in 2023 (n = 24). Study sites 1 and 4 in Fig. 2b-c.

Susineva-site were 5.1 ± 0.6 kg C m^{-2} and 130 ± 2 g N m^{-2}, respectively (Fig. 8). However, the differences between poorly-drained area 1 and well-drained area 2 were large. In area 1, the total C and N loss in the surface peat were 2.3 ± 0.2 kg C m^{-2} and 44 ± 3 g N m^{-2}, respectively. In area 2, the total C and N losses were 5.6 ± 0.2 kg C m^{-2} and 145 ± 5 g N m^{-2}, respectively.

Results of the peatland basin inventories in years 1999 and 2023 in the Susineva-site are shown in Table 4. The total C and N stores of both research areas 1 and 2 had declined. The area weighted average for C and N loss for the entire peatland basin were 5.0 ± 0.05 kg C m^{-2} and 118 ± 1 g N m^{-2}, respectively (Fig. 8). Similar to the surface peat mass loss estimation, the differences between poorly-drained area 1 and well-drained area 2 were large. In area 1, the average C and N loss were 2.9 ± 0.2 kg C m^{-2} and 109 ± 7 g N m^{-2} yr^{-1}, respectively. In area 2, the average C and N losses were larger, 5.4 ± 0.4 kg C m^{-2} and 119 ± 9 g N m^{-2} yr^{-1}, respectively.

Overall, based on these two different approaches, the average area weighted C loss for area 1 and 2 was estimated at 2.6 ± 0.1 and 5.5 ± 0.3 (SE) g C m^{-2}, respectively. The corresponding N loss was 83 ± 10 and 123 ± 7 (SE) g N m^{-2}, respectively.

Peatland margins were not the focus of comprehensive research; however, the relatively shallow peat deposits along the edges and around mineral soil islands within the peatland had burned most intensively (Fig. 7a–g), with observed peat losses of 30–40 cm in thickness. Applying the average surface peat dry bulk density of 71.4 ± 3.0 kg m^{-3} (n = 107) reported for drained ombro–oligotrophic forested

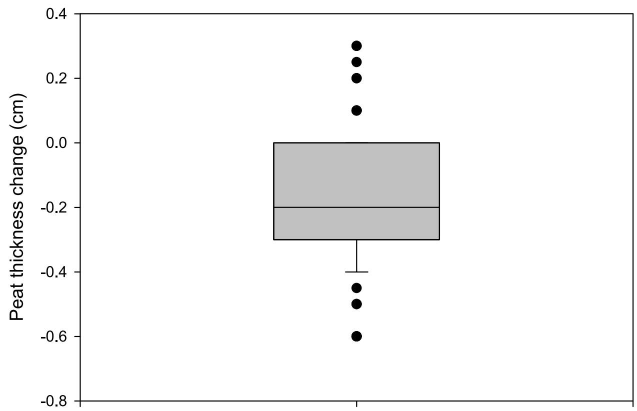

Fig. 6. Box plot showing the changes (cm) in individual peat thickness measurements (n = 108) between the 2023 and 1999 inventories in Susineva peatland site. The whiskers and box bands represent the minimum, first quartile, median, third quartile, and maximum values. The average change in peat thickness is 18 cm.

Table 4
Characteristics (mean ± SE) of the poorly- (area 1) and well-drained (area 2) sites in the Susineva peatland for the years 1999 and 2023. Dry bulk density values represent the mean for the entire peat profiles. 1 Gg = 10^9 g.

Study year / area	Area ha	Average thickness m	Dry bulk density kg m^{-3}	Peat mass mill. m^3	Peat carbon mass Gg	Peat nitrogen mass Gg
1999						
1	13.4	2.14	80.6 (6.4)	0.287	12.5 (0.04)	0.47 (0.00)
2	76.4	1.84	86.7 (6.5)	1.406	67.0 (0.63)	1.48 (0.01)
2023						
1	13.3	2.02	80.4 (5.0)	0.278	12.1	0.46
2	75.8	1.69	86.8 (7.1)	1.320	62.9	1.39

bogs (Turunen et al., 2024), which are representative of the peatland types found in Susineva, the estimated C and N losses in these margin areas was 10–15 kg C m^{-2} and 240–350 g N m^{-2}, respectively.

4. Discussion

4.1. Estimating carbon and nitrogen loss

Estimates of total C and N loss during single fire events on drained boreal peatlands are limited. In this study, detailed pre- and post-fire peatland inventories made in Susineva allowed us to estimate the impact of fire on the C and N stores of a large forestry-drained peatland in Finland. For the extensive forestry-drained peatland area (76 ha, referred to as area 2), the estimated average C loss due to fire (5.5 kg C m^{-2}) exceeded the typical range of 1.5–4.0 kg C m^{-2} reported for boreal undrained peatlands or upland forests (Kuhry, 1994; Pitkänen et al., 1999; Robinson and Moore, 2000; Turetsky and Wieder, 2001; Boby et al., 2010; Turetsky et al., 2011b). In contrast, within a smaller, poorly-drained section (13 ha, referred to as area 1) of the same peatland complex, the average C loss (2.6 kg C m^{-2}) fell within the range observed for boreal undrained peatlands.

The observed C loss in the well-drained Susineva-site (5.5 kg C m^{-2}) and depth of burn (15 cm) were comparable to findings from heavily drained experimental peatlands in Canada, where ditch spacing of 9 m resulted in C losses ranging from 4.7 to 6.7 kg C m^{-2} and a burn depth of 16 cm (Wilkinson et al., 2018). However, these values were substantially higher than those reported for moderately drained Canadian peatlands with wider ditch spacing (18 m), which exhibited lower C losses of 1.7–2.4 kg C m^{-2} (Wilkinson et al., 2018). Despite Susineva's considerably wider ditch spacing of 40 m, which is typical of Finnish drained peatlands, our results suggest that high-severity burning can occur across various drainage intensities and over large areas, rather than being confined to margins. This finding is particularly significant for larger peatlands, where margins constitute a smaller proportion of the total area (Mayner et al., 2018), highlighting the widespread vulnerability of drained peatlands to fire.

Recent modelling by Wilkinson et al. (2023) estimates high average C emissions (10–25 kg C m^{-2}) from wildfires in degraded boreal and temperate peatlands, similar to the findings of Turetsky et al. (2011a), who reported mean C losses of approximately 17 kg C m^{-2} from drained fen plots in western Canada. These values exceed our estimates for boreal forestry-drained peatlands but are more comparable to reported C losses from boreal mire margins and tropical peatlands. For instance, C losses of up to 27 kg C m^{-2} have been recorded from boreal mire margins (Lukenbach et al., 2015; Hokanson et al., 2016), while tropical peat soils in Indonesia have reported losses of 10–23 kg C m^{-2} (Page et al., 2002; Rodríguez Vásquez et al., 2021). The observed differences in C loss rates across various peatland types can be attributed to multiple factors, including variations in dry bulk density, burn depth, peatland size, climate, anthropogenic influences, and hydrogeological conditions (Turetsky et al., 2015; Lukenbach et al., 2015; Mayner et al., 2018; Wilkinson et al., 2023).

The average N emissions in our study (123 g N m^{-2}) align closely with the 30–140 g N m^{-2} range reported by Boby et al. (2010) for boreal forests. However, the importance of N losses in vegetation and organic soil fires is difficult to evaluate because the composition of N emissions depends on several factors, including the vegetation type, C:N ratio, the temperature of the fire, the moisture content and the availability of oxygen during combustion (Torres-Rojas et al., 2020). In laboratory experiment, around one- to two-thirds of the total N is lost upon biomass combustion and the emitted N gas in peat is dominated by ammonia (NH_3), hydrogen cyanide (HCN), a potent greenhouse nitrous oxide (N_2O) and nitrogen oxides (NO_x) (Watson et al., 2019). Overall, however, the variation may be large in actual field conditions and further research is needed to evaluate the impact of N emissions from peatland fires.

4.2. Reliability of the data

4.2.1. Subsidence and burn depth

Does subsidence play a role in Susineva between the peatland inventories of 1999 and 2023? Generally, peatland drainage leads to surface subsidence due to peat shrinkage and biological oxidation, particularly in cultivated and tropical peatlands (Hooijer et al., 2012; Ikkala et al., 2021). Subsidence rates are largely influenced by the water table level (Evans et al., 2019) and vary globally, ranging from millimeters per year in boreal regions to centimeters per year in tropical areas.

In forestry-drained peatlands like our study sites, subsidence tends to occur rapidly after drainage, then slow over time (Minkkinen and Laine, 1998; Sloan et al., 2019). Minkkinen and Laine (1998) found that after 60 years, average subsidence in boreal Finland was only 22 ± 17 cm, with a trend of decreasing subsidence toward the north. Sloan et al. (2019) observed a similar pattern in temperate UK: an initial rapid subsidence followed by minimal change over 20 years. These studies suggest that physical changes in peat after drainage, rather than oxidation, are the main drivers of subsidence in boreal regions.

Susineva was primarily drained in the late 1960s–1970s with 40-meter ditch spacing and has since developed young Scots pine stands without harvesting. The first site inventory in 1999 occurred about 30 years post-drainage, suggesting that the major subsidence has happened during the initial 30 years, with minimal subsidence since then. This was

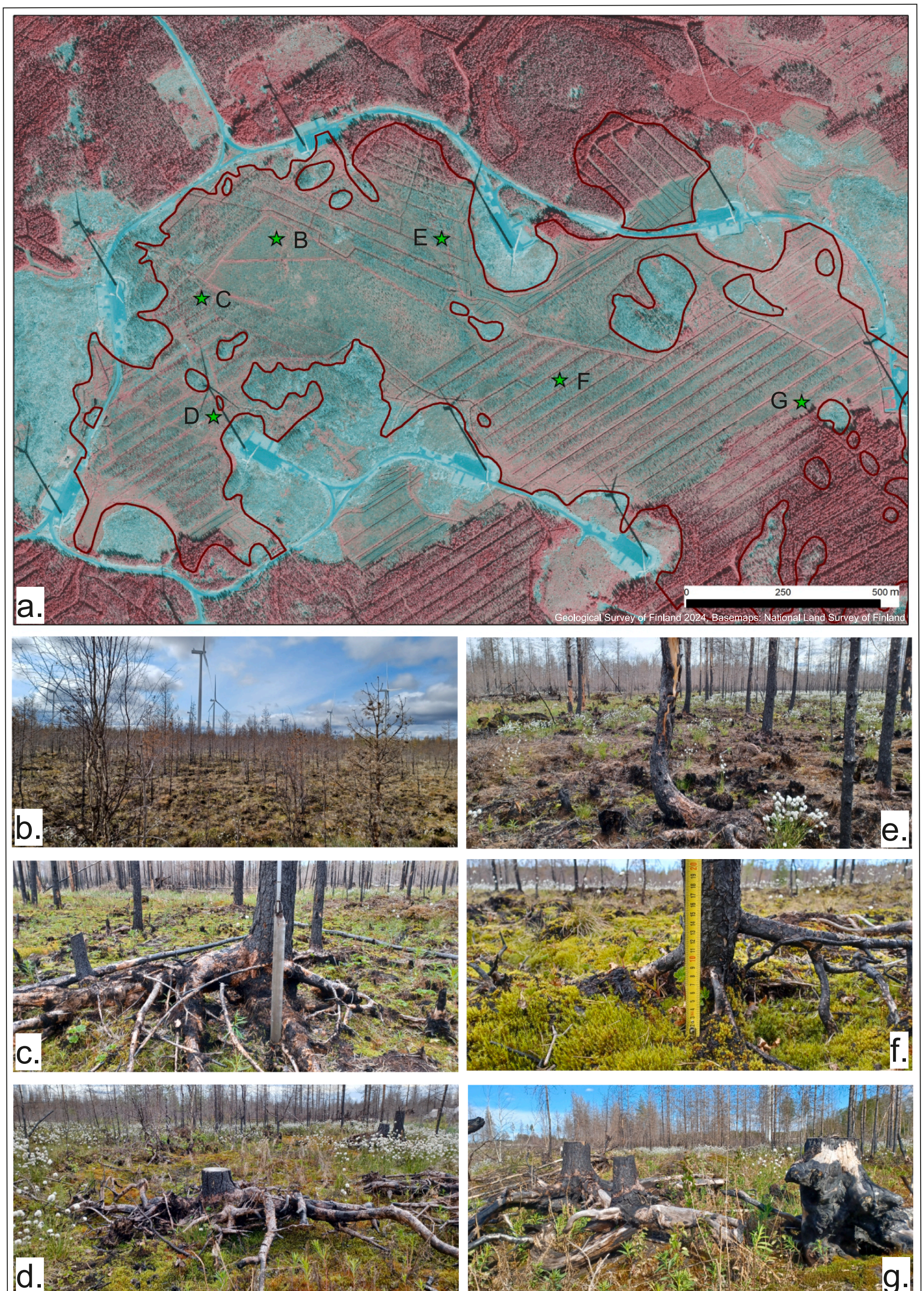

Fig. 7. False color image of Susineva peatland site (a) in which the photos of the different sites (b – g) have been indicated. Photos: J. Turunen and J. Palola.

supported by peat thickness measurements from 1999 and 2023, which closely match observed burn depths (Fig. 7). Also, an increase in thickness was noted at nine study sites. These sites were primarily located in the less intensively drained Area 1, which experienced fewer burns. The observed increase in thickness at these sites could be due to peat accumulation in hummocky areas or errors in locating the exact spot of the 1999 survey. Nevertheless, these sites were included in the analysis equally with the other thickness sites. At the Muhos site, where pre-drainage peat thickness was under 100 cm, subsidence is also likely minimal. According to Minkkinen and Laine (1998), initial post-drainage subsidence correlates strongly with peat thickness, being lowest in peatlands with shallow peat deposits. Overall, it looks likely that subsidence has played a minimal role at both Susineva and Muhos sites over the last two decades.

In Susineva, the accuracy of the results could have been further verified using the root method (Kasischke et al., 2008). However, the

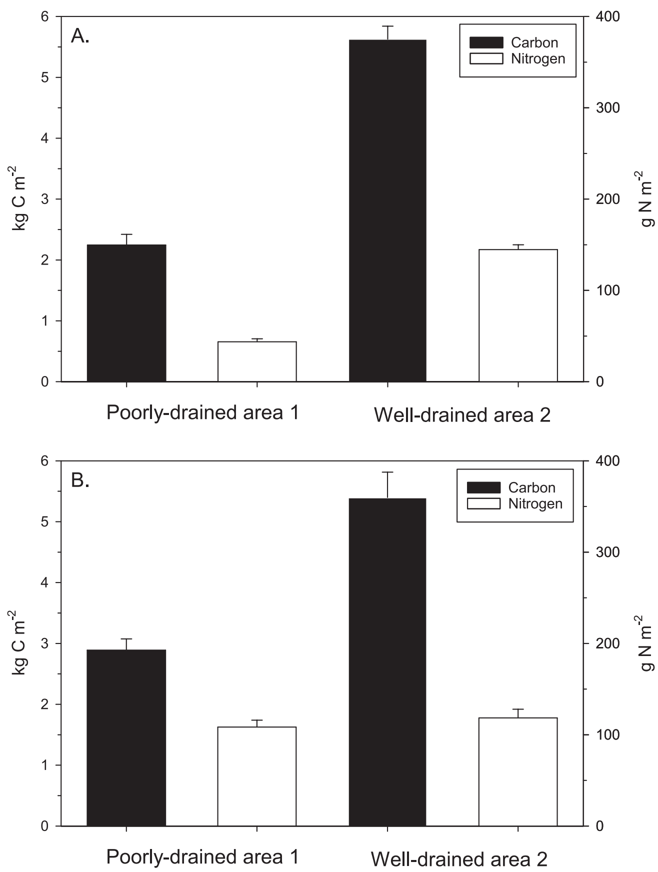

Fig. 8. The average carbon (g C m^{-2}) and nitrogen (g N m^{-2}) loss of poorly- and well-drained sites in the Susineva peatland: (A) based on the surface peat mass estimation, and (B) the whole peatland basin inventory.

method for estimating burn depth is also subject to several uncertainties, primarily due to challenges in determining the pre-fire organic layer depth. It relies on the position of adventitious roots as a proxy for pre-burn soil depth, but this introduces variability because root position can vary across tree ages and site conditions. Additionally, spatial heterogeneity in organic layer thickness and burn severity, coupled with limitations of sampling methods, reduce the accuracy and representativeness of burn depth estimates across the landscape (Kasischke et al., 2008). Deeper burning beneath tree canopies can be a source of error, attributed to drier fuel material resulting from greater interception by the canopy (Miyanishi and Johnson, 2002). Considering all these uncertainties, we believe our estimates remain robust and reliable.

4.2.2. *Accuracy of dry bulk density estimates*

Accurate dry bulk density estimates are crucial for calculating C and N losses, as variability in these values can introduce significant errors. In Susineva, estimates of dry bulk density for weakly decomposed (H_{1-3}) *Sphagnum* surface peat were relatively consistent across two methods used in the 1999 inventory: 32 and 45 kg m^{-3} for poorly drained sites, and 71 and 78 kg m^{-3} for well-drained sites. To account for uncertainty, we applied the standard error of surface peat dry bulk density estimates from Turunen et al. (2024) to our C and N loss calculations. This resulted in estimated error margins of 13 % for the poorly drained site (Area 1) and 5 % for the well-drained site (Area 2). These error margins indicate that the dry bulk density values used in this study provide a reasonably reliable basis for estimating C and N losses. In the forested peatlands of the Muhos site, which have dense stands originating from open *Carex* fens, the condensed surface layers exhibit minimal variation. Similarly, undrained *Sphagnum* bogs show uniform, weakly humified surface layers. Given that the standard error estimates for Muhos site are of the same magnitude as those used for Susineva, the surface dry bulk density values used for Muhos can also be considered likely accurate and represent the best available estimates for these peatland types.

Drained peatlands of Muhos site with sparse tree cover, in contrast, show greater uncertainty due to varied drainage histories. However, surficial *Sphagnum* growth across all sites suggests that average values for *Sphagnum* peat in drained peatlands remain valid. Unstable mires pose specific challenges due to contrasts between loose *Sphagnum*-dominated top layers and dense *Carex* layers below (Laitinen, 2008). Burn depth variation can be significant, but mean values for *Carex* peat in undrained mires still offer the best magnitude estimates.

In upland forests, the main uncertainty lies in the humus layer thickness. Average values for *Cladonia*-type forests may overestimate conditions at Muhos, where older fires have reduced the humus layer, though this inaccuracy is minor.

4.2.3. *Carbon and nitrogen loss by drainage*

C loss estimates from two approaches in our study are consistent. However, the whole-basin method likely includes C and N losses from drainage-related CO_2 emissions and runoff of dissolved organic carbon (DOC) and nitrogen. In boreal forestry-drained peatlands, C losses are higher in nutrient-rich areas (Ojanen et al., 2013). Drainage alters moisture levels, increasing CO_2 emissions, DOC and N runoff, and reducing CH_4 emissions (Alm et al., 2007; Nieminen et al., 2018, 2021; Rantakari et al., 2010; Sallantaus, 1992; Turunen and Valpola, 2020).

In low-fertility peatlands like Susineva, combined CO_2 and DOC losses are estimated at 7–10 g C m^{-2} yr^{-1} (Ojanen et al., 2013; Sallantaus, 1992; Rantakari et al., 2010). N export is estimated at 0.10–0.18 g N m^{-2} yr^{-1} (Nieminen et al., 2020). Over the last 20 years, total C and N losses from drainage may account for about 3–5 % of the total C loss. Overall, this study demonstrates the profound impact of a high-intensity fire event on C dynamics and storage within forestry-drained peatlands. The release of peat C into the atmosphere during a fire event was found to be approximately 500 to 800 times greater than the annual combined CO_2-C and dissolved organic carbon (DOC) loss typically observed in these drained, low-fertility peatlands. Additionally, N emissions from fires were even higher, approximately 700 to 1200 times higher than the estimated annual N export.

4.3. *The vulnerability of different environments to fires*

Although the burn severity in Susineva was extensive, disturbance patterns were spatially heterogeneous across the study site. The most intense burning occurred in areas with relatively shallow peat deposits, especially along peatland margins and near mineral islands (Fig. 7a–g), consistent with the findings of Wilkinson et al. (2020). However, large C losses have been reported across a range of peat thicknesses, with reported mean values varying widely from 1.6 to 27 kg C m^{-2} (Lukenbach et al., 2015; Hokanson et al., 2016; Mayner et al., 2018; Wilkinson et al., 2020). In our study, the highest estimated C losses (10–15 kg C m^{-2}) were observed at several margin sites, falling approximately in the midrange of previously reported values and being approximately double the average in other forestry-drained areas of Susineva.

Hydrologically unstable mires, such as those near sandy esker complexes in the Muhos area, are particularly vulnerable to wildfires, especially in early stages of peat development when peat layers are thin (<30 cm) and unable to buffer water table fluctuations. Our results from the Muhos-site, which has experienced two large fires in the past 50 years (1969 and 2020), show that undrained unstable mires can lose their entire surface peat layer during fire events, with C losses reaching 3.6 kg C m^{-2}, significantly higher than in undrained *Sphagnum*-dominated mires (0.25 kg C m^{-2}), though lower than in drained forested peatlands (5.3 kg C m^{-2}). Pitkänen et al. (1999) similarly found that numerous fires in Patvinsuo mire in eastern Finland slowed peat growth, particularly during the early stages of mire development when the peat layer was thin (<20 cm), with an estimated C loss of 2.5 kg C m^{-2} per combustion event. This is comparable to the peat C loss observed in the undrained unstable mires of Muhos. Similar patterns of fire-driven peat loss and slowed accumulation have been observed across Finland, including Lakkasuo, southern Finland (Pitkänen et al., 2001), where fire activity was concentrated in basal layers and along mire edges, underscoring the long-term impact of fire on C dynamics in peatlands with unstable hydrology.

Most undrained mires are two-layered systems (Ingram, 1978), with the acrotelm facilitating lateral water flow and a vertical gradient in hydraulic conductivity that limits water loss. This structure keeps the deeper peat layer (catotelm) continuously waterlogged, preventing the surface from fully drying. However, mires near permeable sandy eskers, like those in Muhos, are often horizontal mires (Joosten et al., 2017) with minimal lateral water movement. These sites experience strong water table level (WTL) fluctuations due to sandy, permeable bottoms, leading to faster decomposition, highly humified peat layer, and reduced water storage capacity (e.g. Huttunen, 1987; Laitinen et al., 2008). Such instability increases fire risk during dry periods, as surfaces dry quickly, and thin peat layers can burn deeply. Vegetation gradients form due to moisture variability (Laitinen et al., 2008), limiting moss growth but supporting certain mire plants that thrive with less competition. In contrast, mires in areas with impermeable bottom soils (Succow and Joosten, 2001) can develop into *Sphagnum* bogs with thick peat layers and stable, moist surfaces.

Approximately 83 % of the estimated C emissions in the Muhos-site originated from peatlands, which cover only about half of the area, with around 94 % of these emissions coming from peat and humus layers. The inventory method used allows only rough comparisons between site types, but drained forested peatlands—especially those with intensive drainage and high stand stocking—produced the most fire-induced C emissions. Mires with variable water regime, even when undrained, can produce similar areal C emissions but cover smaller areas. Acrotelm mires, particularly hummock-level bogs, can act as barriers to fire spread. The inventory showed that the burned acrotelm mires saw some surface peat loss in flark and lawn levels, but hummocks remained largely intact. Further, especially in undrained burned peatlands with variable hydrological conditions, fires led to the formation of shallow water ponds that support numerous breeding birds, while also enabling the post-fire expansion of the threatened moss *Sphagnum molle* carpets spreading over mud bottoms. Overall, fires increased biodiversity in undrained mires, whereas drained peatlands experienced elevated greenhouse gas emissions without corresponding ecological benefits.

4.4. Vulnerability of forestry-drained peatlands to fires

Compared to undrained boreal peatlands, forestry-drained peatlands are generally more vulnerable to fire disturbance due to large ecohydrological effects of forest drainage practices (Wilkinson et al., 2018, 2023). The drainage of peatlands lowers the WTL and isolates them from groundwater flow systems leading to soil degradation processes such as subsidence and increased thickness of the aerated surface layer. This in turn increase the surface peat dry bulk density through microbiological activity, mineralization and decomposition, resulting in progressive loss of peat forming mire species like *Sphagnum* mosses. Overall, these processes deteriorate the peat-forming conditions and thereby diminish ecosystem resilience (e.g. Laine et al., 1995a,b; Sherwood et al., 2013; Kettridge et al., 2015; Wilkinson et al., 2018, 2023). Additionally, drained peatlands undergo pronounced shifts in vegetation composition towards forest communities (Laine et al., 1995a,b; Jauhiainen et al., 2002; Punttila et al., 2016). The increased woody vegetation and litter (shrubs, tree stand) can promote the spread of fire during periods of hot and dry conditions (Higuera et al., 2009; Kettridge et al., 2015). Combined with the growing frequency of drought events linked to climate change, these vegetation changes heighten the risk of both increased frequency and severity of peatland fires in the boreal region (Kettridge et al., 2015).

In less intensively drained areas with low-stocked tree stands (area B in Muhos and area 1 in Susineva), C and N losses were 40–70 % lower than in more intensively drained areas. This difference can be attributed to sparser drainage network and lower forest productivity resulting in larger soil moisture content and smaller amount of above-ground fuel material compared to drained peatland forests. Also, our surface peat measurements from the Susineva-site showed a water content approximately 10 % higher (91 %) in poorly-drained and less intensively burned area than the rest of the peatland (85 %), indicating a higher WTL and consequently, greater ecohydrological resilience (Sherwood et al., 2013).

In the peatland forest areas like Muhos (area A) and Susineva (area 2), dense young Scots pine stands and increased plant litter biomass likely enhanced fuel availability and created drier surface conditions, promoting fire spread. This is supported by evidence of deeper burning beneath tree canopies, which has been attributed to drier fuel material resulting from larger interception from the canopy (Miyanishi and Johnson, 2002) or reduced latent heat sink during propagation of the combustion front associated with tree roots and stems (Greene et al., 2007; Boby et al., 2010). In Fennoscandia, the probability of forest fire frequency has increased over the last 50 years (Niklasson and Granström, 2000; Schimmel and Granström, 1997), now averaging a 40–50 year return interval (Niklasson and Granström, 2000; Pitkänen et al., 2001). This increase is attributed to the accumulation of fuels during forest succession. However, while most fires occur on upland mineral soils, peatland fires remain rare (Lindberg et al., 2021). In Finland, the drainage of peatland forests, mostly conducted between 1960 and 1980, has significantly increased forest stand productivity, resulting in current stand volumes and densities that increase fire probability. On peatlands, this fuel-fire feedback mechanism may be more significant than in upland forests. Trees also dry their substrate through evapotranspiration (Sarkkola et al., 2010) and as the stand grow, its evapotranspiration capacity also increases, leading to larger and deeper drying of surface peat. This can increase the ignition probability as well as fire severity and depth. These characteristics of peatland forests should be incorporated into future fire risk models.

It is notable that the Kalajoki fire was a relatively large-scale fire event covering 290 ha and the entire peatland area. In undrained peatlands, observations of charcoal layers in Finland suggest that few, if any, fires have spread over an entire peatland basin during its whole Holocene history (Tolonen and Ruuhijärvi, 1976). Although Finland is experiencing approximately 1000 separate forest fires annually, the average fire size remains small, typically less than 1 ha, due to active and effective fire suppression efforts (Lehtonen et al., 2016). Similarly in Canada, fires larger than 200 ha represent only 3 % of the total number of fires but account for majority of the area burned (Stocks et al., 2002). In 2023, there were 834 such large fires in Canada, more than 2.5 times the 1986–2022 average, and 60 very large fires (each > 50 000 ha) were responsible for 73 % of the total burned area (Jain et al., 2024). This pattern aligns with long-term fire regime trends, where a small number of large fires disproportionately drive annual burned area (Hanes et al., 2019; Jain et al., 2024). Also, smouldering fires are becoming an increasing global concern due to their potential for significant C loss and the challenges they pose in detection and suppression (Lukenbach et al., 2015; Rein and Huang, 2021).

Overall, the fire intensity is highly sensitive to both weather and climate conditions and the developmental stage of the forest stand. With climate change, the anticipated increase in both the fire frequency and burn severity within the boreal zone will significantly impact the cycling and storage of terrestrial C and N (Wilkinson et al., 2023). However, on forestry-drained peatlands, the ecological and economic impact can be larger compared to undrained peatlands because of their significantly higher value for wood production. The results of this study underscore the importance of recognizing the ecosystem service benefits, particularly in the context of management-related drivers such as drainage and fires, which can collectively exert severe effects on the peatland C balance and overall vulnerability, including reduced fire resilience.

It is probable that the impact of fires will continue depleting the C and N stores both on undrained and drained boreal peatlands together with human-induced net CO_2 soil emissions (e.g. Turetsky et al., 2002, 2015; Kettridge et al., 2015; Wilkinson et al., 2023). Moreover, the results highlight the critical importance of adopting sustainable management practices in peatland forestry, especially regarding drainage, harvesting methods, and tillage, which would decrease the flammability and the burning depth in the peatland forests. This need is further reinforced by the extensive drying of European peatlands in recent centuries (Swindles et al., 2019). Vital strategies to mitigate the negative emissions resulting from drainage and fires include enhancing moisture conditions in drained peatlands by maintaining higher water table levels also in drought periods. In nutrient-poor and unproductive forestry-drained peatlands, rewetting such as mire restoration can reverse the degradation processes and mitigate the effects of drainage on biodiversity and C sequestration, thereby fostering increased fire resilience of the peatland but also the adjacent upland forest sites if the restored areas are sufficiently large, wet and uniform to form barriers between forest areas.

CRediT authorship contribution statement

Jukka Turunen: Writing – review & editing, Writing – original draft, Visualization, Project administration, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Sakari Rehell:** Writing – review & editing, Writing – original draft, Visualization, Validation, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Sakari Sarkkola:** Writing – review & editing, Writing – original draft, Validation, Methodology, Formal analysis, Conceptualization. **Harri Vasander:** Writing – review & editing, Writing – original draft, Validation, Project administration, Methodology, Funding acquisition, 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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Data availability

Data will be made available on request.

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