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Impacts of thinning activities on boreal peatland forests

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Academic dissertation

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ABSTRACT

Boreal peatland forests are an important source of timber. Recently, timber harvesting has been extended to warmer months, resulting in machinery traffic over unfrozen soils, and leading to higher levels of soil disturbance, such as deeper ruts. Despite this, our knowledge of the impact of soil disturbance on peat physical properties and soil biochemistry is still limited. To address this gap, I conducted a study to examine the effects of soil disturbance caused by harvesting machinery during thinning operations on the soil physical, chemical, and biological properties and vegetation of drained boreal peatland forests. To assess the rate of recovery, I sampled six sites that formed a chronosequence covering 15 years since thinning. The results showed that soil disturbance caused an increase in the bulk density and field capacity of peat, along with a decrease in total porosity. In the vegetation, moss biomass and root production were reduced, but sedge cover increased. Furthermore, recently disturbed areas exhibited greater soil CO_2 production potential, as well as higher soil CO_2 and CH_4 concentrations compared to control areas. However, CO_2 and CH_4 emissions, microbial communities, and cellulose decomposition rate were not impacted. Although the rate of recovery varied, all studied properties impacted by disturbance were fully recovered within 15 years. As the water retention characteristic (WRC) describes soil structure and its alterations, it a useful for disturbance assessment. Thus, I propose how WRC can be predicted using artificial neural networks. Overall, the study demonstrated that while drained boreal peatlands are sensitive to disturbance, they are also resilient to mechanical soil disturbance caused by thinnings.

Keywords: water retention, pore size distribution, soil CO2, CH4 and N2O concentrations, soil CO2 and CH4 emissions, biomass, decomposition

Boreaaliset suometsät ovat tärkeä puunlähde. Viimeaikainen puunkorjuun laajeneminen sulan maan aikaan, on lisännyt maaperälle aiheutuvia häiriötä, kuten syvien korjuu-urien muodostumista. Tästä huolimatta korjuun aiheuttaminen häiriöiden vaikutuksista turpeen fysikaalisiin ominaisuuksiin ja maaperän biokemiaan tiedetään vähän. Tämän puutteen korjaamiseksi tutkin puunkorjuukoneiden harvennustöiden aikana aiheuttamien maaperän häiriöiden vaikutuksia ojitettujen boreaalisten suometsien kasvillisuuteen ja maaperän fysikaalisiin, kemiallisiin ja biologisiin ominaisuuksiin. Palautumisnopeuden arvioimiseksi otin näytteitä kuudelta alalta, jotka muodostivat aikasarjan, joka kattoi 15 vuotta harvennuksesta. Tulokset osoittivat korjuun lisäävän turpeen tiheyttä ja vedenpidätyskykyä sekä pienentävän kokonaishuokoisuutta. Kasvillisuuden sammalbiomassa ja juurituotanto vähenivät, mutta sarojen peittävyys lisääntyi. Lisäksi äskettäin häirityillä aloilla oli suurempi maaperän CO_2-tuotantopotentiaali sekä korkeammat maaperän CO_2 - ja CH_4-pitoisuudet verrattuna kontrollialueisiin. Korjuu ei kuitenkaan vaikuttanut CO_2 - ja CH_4-päästöihin, mikrobiyhteisöihin ja selluloosan hajoamisnopeuteen. Vaikka palautumisnopeus vaihteli, kaikki tutkitut ominaisuudet olivat täysin palautuneet 15 vuoden kuluttua. Koska vedenpidätysominaisuus (WRC) kuvaa maaperän rakennetta ja sen muutoksia, sitä voidaan käyttää häiriöiden arviointiin. Tutkimuksessani osoitan kuinka WRC:n voidaan arvioida keinotekoisten neuroverkkojen avulla. Kaiken kaikkiaan tutkimus osoitti, että vaikka ojitetut boreaaliset suot ovat häiriöherkkiä, ne myös palautuvat hyvin harvennusten aiheuttamista mekaanisista maaperän häiriöistä.

Avainsanat: vedenpidätys, huokoskokojakauma, maaperän CO_2-, CH_4- ja N_2O-pitoisuudet, maaperän CO_2- ja CH_4-päästöt, biomassa, hajotus

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Jyväskylä, November 2023
Dmitrii Lepilin

LIST OF ORIGINAL PAPERS

The thesis is based on two published articles and one manuscript which are cited in the text as Roman numerals **I** to **III**. Articles **I**, and **II** are reprinted with permission of the Canadian Journal of Forest Research while submitted manuscript **III** represents the author's version.

I. **Lepilin D,** Laurén A, Uusitalo J, Tuittila E-S (2019) Soil deformation and its recovery in logging trails of drained boreal peatlands. Can. J. For. Res. 49(7): 743–751. Canadian Science Publishing. doi:10.1139/cjfr-2018-0385.

II. **Lepilin D,** Laurén A, Uusitalo J, Fritze H, Laiho R, Kimura B, Tuittila E-S (2022) Response of vegetation and soil biological properties to soil deformation in logging trails of drained boreal peatland forests. Can. J. For. Res. 52(4): 511–526. NRC Research Press. doi:10.1139/cjfr-2021-0176.

III. **Lepilin D,** Laiho R, Laurén A, and Tuittila E-S (2023) Artificial neural networks for predicting soil water retention characteristic of boreal peatlands. Submitted manuscript.

Author's contribution

This thesis has been compiled by: D. Lepilin (DL). DL was the first author of articles I, II and manuscript III with contribution of the co-authors. All authors including DL participated in planning of the study. Statistical analysis, visualization, and modeling were executed by DL with contribution of E.-S. Tuittila and A. Laurén.

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1. INTRODUCTION

The boreal zone includes a significant share of forested peatlands, which accounts for almost 24% of the forest-covered area (Wieder et al. 2006). Most of those peatlands were drained in order to improve tree production. Generally, drainage has a negative impact on other ecosystem services than timber production, such as soil carbon storage and water regulation. The soil carbon storage service of peatlands is of particular importance as it significantly contributes to the global greenhouse gas cycle. That is makes an especially strong case for the restoration of previously drained peatlands. However, peatlands used for forestry are still playing a significant role in the economy of northern countries, e.g. such as Finland where they account for 26% of the overall forest-covered area (Päivänen 2008). Forested peatlands and upland forest forests grown on those peatlands are managed using the same forestry practices as in upland forests, including harvesting with heavy machinery (Päivänen and Hånell 2012).

It is well documented that heavy machinery harvesting, performed in upland forests, leads to soil deformation reflected in compaction and rut formation, structural changes of microbial communities, and root damage (Frey et al. 2011; Hartmann et al. 2014; Host 1996; Nawaz et al. 2012). Nonetheless, peat differs from mineral soils. The main difference is its predominant organic nature (organic content > 75%) which is reflected in peat’s ability to hold a higher amount of water, high compressibility, and low shear strength (Päivänen and Hånell 2012). Those characteristics are reflected in the lower bearing capacity of peat, which is one of the main reasons for winter harvest of peatland forest while the soil was frozen, which was earlier a common practice. Although the negative impact is not entirely eliminated (Groot 1987), it is significantly reduced in comparison to harvesting on unfrozen peat. However, global warming and increasing demand for timber force a shift of harvesting operations to be done during warmer periods while peat is unfrozen and therefore susceptible to disturbance (Uusitalo and Ala-Ilomäki 2013).

1.1 Impact of harvesting machinery to peat

Generally, soil disturbances after harvesting are similar on peat and mineral soils. In Canada, those disturbances are a major problem in peatland forestry (Jeglum 1983). The most obvious consequences of harvesting are rutting and subsequent soil erosion (Groot 1987; Locky and Bayley 2007; Nugent et al. 2003; Satrio et al. 2009). However, there are internal changes in soil structure which are not obvious to the naked eye. Primarily, harvesting affects peat’s physical properties, such as bulk density and water retention characteristics (Chow et al. 1992; Nugent et al. 2003). Changes in water retention characteristics impact moisture and aeration dynamics of peat which can have a negative impact on redox and biogeochemistry of soil. The primary drivers of change in water retention characteristics are structural changes in e.g., porosity. Rutting on mineral soils is associated with decreased total porosity (Cambi et al. 2015; Hansson et al. 2018), along with changes in pore size distribution in favor of smaller pores (Hillel 1982). Those changes impact soil drainability through decreased hydraulic conductivity (Hansson et al. 2018). Consequently, slower movement of water decreases air-filled porosity and gas diffusion.

Those changes, depending on their significance, could affect plant communities and soil biological processes as found in undrained peatlands after linear disturbances (Davidson et al. 2021; Echiverri et al. 2020). One of the main reasons for such changes is restricted gas exchange as a result of decreased air-filled porosity. This is expected to increase levels of carbon dioxide (CO_2) in peat, similar to the CO_2 increase found in mineral soils (Gaertig et al. 2002; Goutal et al. 2012). Excessive CO_2 accumulation further alters the soil environment. The above factors were found to limit root growth (Bodelier et al. 1996; Startsev and McNabb 2001), microbial activity (Frey et al. 2009; Marshall 2000), and cause changes to microbial communities (Jordan et al. 2003; Li et al. 2004; Tan et al. 2005) in mineral soils.

The most recent research dedicated to the influence of exploration lines, used to transport natural resources on undrained Canadian peatlands, showed a significant impact on soil properties, vegetation, and biogeochemistry. Disturbances found on those lines resemble post harvesting disturbances and caused increased bulk density, volumetric water content, and decomposition rate (Davidson et al. 2020). Along with changes in peat physical properties, (Davidson et al. 2021) reported phenological change manifested in earlier seasonal peak with decreased moss abundance and vegetation shifted to sedge and willow dominance. Structural changes in vegetation showing replacement of feather moss with *Sphagnum* moss, were also reported by Deane et al. (2020) for exploration lines crossing undisturbed forested peatland. While Echiverri et al. (2020) found a recovery of understory cover towards undisturbed treed fens.

Recovery of mineral soil after similar disturbances may take decades (Cambi et al. 2015; Erler and Güldner 2002; Froehlich et al. 1985; Heninger et al. 2002; Horn et al. 2007), while the share of a disturbed area may take up to 40 % of the harvested site (Grigal and Brooks 1996). Considering the share of the impacted area and the special role of peatlands in the global greenhouse gas cycle it is necessary to improve understanding of the machinery-induced impact on peat physical properties, vegetation, soil biological activity, and biogeochemical cycling as well as their recovery potential.

1.2 Predicting water retention characteristic of peat

Earlier was noted that soil water retention characteristic serves as an integral physical property and its variation is critical for biological and chemical activity through moisture and aeration regime (Goutal et al. 2012). That makes it potentially suitable for the estimation of machinery impact on peat and subsequent recovery. However, the determination of water retention characteristics is expensive and time-consuming. Thus, an accurate prediction model could be a reasonable alternative to direct field and laboratory measurements.

The integral nature of water retention characteristics makes it logical to use the pedotransfer functions (PTF) approach (Bouma 1989). This approach involves the determination of certain soil properties that are laborious to measure based on other easier measurable properties. One commonly used method to derive PTF is the application of artificial neural networks (ANNs) (Van Looy et al. 2017). This method does not need an initial model concept, which makes it universal for function approximation. However, this advantage of ANNs has a drawback in the form of black box nature and following from this non interpretability of developed models.

ANN-based PTFs are successfully used to derive water retention characteristics for mineral soils (e.g., D'Emilio et al. 2018; Haghverdi et al. 2014; Moreira de Melo and Pedrollo 2015). However, they have not been applied to peat soils. Previously, PTFs for peat soil were regression-based (e.g., Weiss et al. 1998), where developed PTFs directly estimate shape parameters for the updated van Genuchten model. ANN-based PTFs have better performance in comparison to regression methods (Minasny et al. 1999; Pachepsky et al. 1996). It is essential to consider that peat is significantly different from mineral soils, which makes it impossible to blindly use the same architecture or same input neurons characterizing soil structure. Instead of soil texture, as in the case with mineral soils, peat soils, due to their organic nature, can be better described by the level of decomposition (estimated using the von Post index) (Parent and Caron 2007).

1.3 Aims of the study

This study aimed to quantify the impact of harvesting machines on the soil and vegetation of drained peatland forests and how their recover after harvesting. As water retention characteristics appeared to be a key factor for the other soil properties, the second aim was to develop a method to estimate water retention characteristics of boreal peatland soils based on easier measurable soil properties. More specifically, the study addressed the following four research questions:

(1) How do harvesting machines impact the physical properties of peat? (**I**)
(2) How do harvesting machines impact vegetation and biological properties of peat? (**II**)
(3) Do peat properties and vegetation recover after the disturbance of harvesting machines? What is the rate of recovery? (**I & II**)
(4) Is ANN modeling using pseudo-continuous PTF suitable for predicting water retention characteristics of boreal peatland soils? (**III**)

2. MATERIALS AND METHODS

2.1 Study design and locations

The research project was carried out at six peatland sites located in southern Finland (Table 1 and Fig. 1). These sites shared several similar characteristics, including the type of peat and tree stand, as well as an average yearly temperature ranging from 4 to 5 ℃. The warmest and coldest months had temperatures of -6.2 to -6.2 ℃ in January and 15.2 to 16.2℃ in July, respectively. The Scots pine (*Pinus sylvestris* L.) was the dominant tree species, and the understory vegetation comprised mainly of forest and peatland dwarf shrubs. The drained peatland forests represented the Vaccinium vitis-idaea (Ptkg) and dwarf shrub (Vatkg) types, based on the classification by Laine and Vasander (2008). These peatlands were drained in the 1960s and 1970s and their ditches were cleaned two decades later. Harvesting activities (thinning) were performed at all sites, resulting in the formation of logging trails and ruts in the top peat layer. These trails and ruts were primarily created during the previous thinning and transportation of harvested timber, with no additional machine traffic afterward. The typical harvester forwarder used had a weight of $15 \cdot 10^3$ to $18 \cdot 10^3$ kg and a maximum carrying capacity of $10 \cdot 10^3$ to $15 \cdot 10^3$ kg.

To evaluate the extent and rate of recovery from harvesting, we grouped the study sites based on the time elapsed since harvesting to form a chronosequence. The chronosequence was divided into three age classes, AC1, AC2, and AC3. AC1 represented conditions immediately following thinning, while AC2 and AC3 represented conditions 4-5 years and 14-15 years after thinning, respectively. The objective was to observe temporal changes in soil properties since harvesting.

We categorized peat disturbance occurring on logging trails based on the depth of the rut. To establish a control, we measured the peat surface unaffected by trafficking, which was located 5 - 10 m away from the trail (referred to as disturbance class DC0). The forest management recommendations designed for peat forests (Vanhatalo et al. 2015) dictate that the maximum depth of logging trails in peatlands should not exceed 0.20 m, which we used as the basis for evaluating disturbance severity. Sites with logging trail depths less than 0.20 m were considered to be moderately disturbed (classified as disturbance class 1, DC1), while those with depths greater than 0.20 m were classified as severely disturbed (classified as disturbance class 2, DC2)."

In studies **I** and **II**, we structured our sites hierarchically by selecting six sites across three age classes. Within each site, we established a total of 12 plots, which were divided into three disturbance classes: six plots in DC0, and three plots each in DC1 and DC2. Overall, the study was composed of 72 plots.

Table 1. Characteristics of the experimental sites. Mean physical properties and total cover of Sphagnum mosses, other mosses, dwarf shrubs, forbs, and sedges are given for the control plots, DC0.

Site	Age class	Average peat depth, cm	Coordinates	Harvesting machinery	Years after thinning	Mean von Post	LOI, %	Mean estimated cover of Sphagnum mosses, %	Mean estimated cover of other mosses, %	Mean estimated cover of dwarf shrubs, %	Mean estimated cover of forbs, %	Mean estimated cover of sedges, %
Varsapuro	AC1	176	lat. 62.60928 lon. 24.62267	8-w Pon. Fox 10-w Pon. Buff.	< 1	3.8 ±0.7	94.4 ±2.2	35 ±37	15 ±17	15 ±10	2 ±4	1 ±1
Permisuo	AC1	98	lat. 62.20058 lon. 24.52608	ProSilva 910 ProSilva 15-4ST	< 1	3.9 ±0.8	96.4 ±1.7	32 ±40	34 ±27	25 ±6	4 ±8	1 ±1
Vuorijärven	AC2	101	lat. 61.82745 lon. 24.3169	J.Deere 1070D J.Deere 810E	4	3.7 ±0.8	97.9 ±0.8	0 ±0	50 ±22	21 ±10	8 ±12	1 ±2
Mustakeid as	AC2	202	lat. 61.76302 lon. 22.64543	J.Deere 1270D J.Deere 1110D ProSilva 15-4ST	5	3.8 ±0.6	97.7 ±1.4	16 ±39	48 ±33	9 ±9	11 ±9	1 ±2
Isoneva	AC3	53	lat. 61.9425 lon. 22.94602	NA	14	4.0 ±0.9	98.1 ±0.9	0 ±0	42 ±26	17 ±5	12 ±27	4 ±7
Vehkasuo	AC3	108	lat. 61.7848 lon. 23.99262	NA	15	3.8 ±0.6	96.6 ±0.8	25 ±38	49 ±37	15 ±14	5 ±4	5 ±7

Note: Disturbance classes: DC0: undisturbed; DC1: rut depth < 0.2 m; DC2: rut depth > 0.2 m.
von Post, peat decomposition stage in the top 10 cm layer according to the von Post scale.
LOI, peat loss on ignition (4 h at 600 °C)
± standard deviation

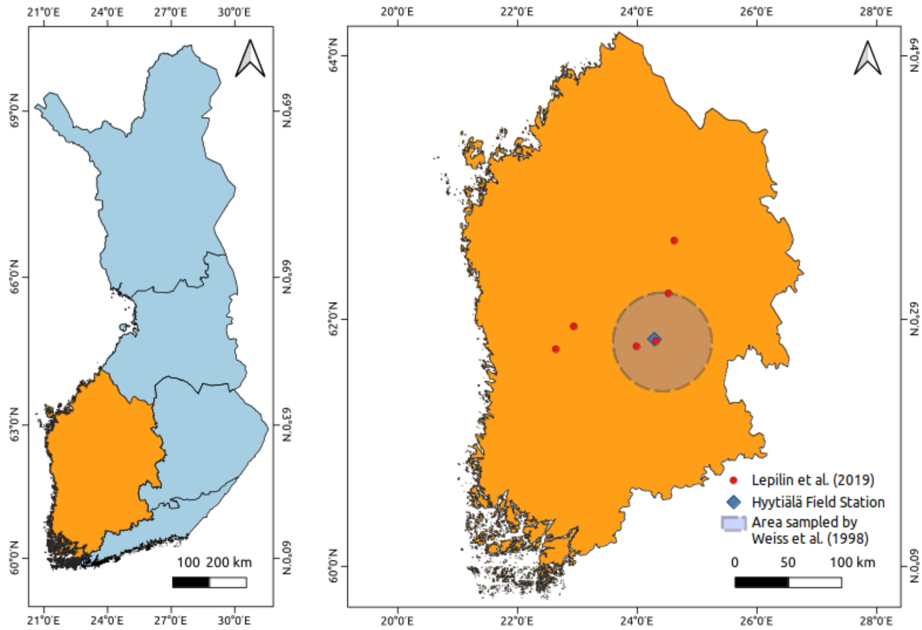


Figure 1. Location of experimental sites and sampling areas within used data sets. Sampling for Päivänen (1973) was done around Hyytiälä field station. Age classes: AC 1: <1 year since thinning; AC2: 4–5 years since thinning; AC3: 14–15 years since thinning. Each experimental site had plots classified according to Disturbance classes: DC0: undisturbed; DC1: rut depth < 0.2 m; DC2: rut depth > 0.2 m.

2.2 Soil physical and chemical properties

To quantify soil properties, we collected volumetric peat cores from 72 sampling plots using a 6×6×60 cm sampler (Jeglum et al. 1991). Various physical parameters were measured in the laboratory, including bulk density (ρ), water retention characteristics, degree of decomposition, and loss on ignition. These measurements were then used to calculate porosity (ϕ_f) and pore size distribution. Bulk density was determined by dividing the oven-dry mass by the initial volume of the sample. To assess the degree of decomposition, the von Post scale (von Post 1922) was utilized, which assigns values ranging from 1 to 10 to indicate the degree of decomposition (Parent and Caron 2007). We determined organic matter content by the loss on ignition method (LOI), while pH was determined from a mixture of 10 ml of fresh peat and 30 ml of distilled water. We used the loss on ignition method (LOI) to determine the organic matter content, and for measuring the pH, we used a mixture of fresh peat in distilled water. The water retention characteristic was determined using the pressure plate extraction method (Reynolds and Topp 2007) and fitted with the van Genuchten-Mualem equation (Mualem 1976). The equation was separately fitted for each sample. A detailed description of the sampling and laboratory analysis of soil samples is given in **I**.

2.3 Vegetation

The vegetation composition, living moss biomass, and root production rate in each of the 72 study plots were analyzed to evaluate the changes in vegetation after disturbance. The method described by Kokkonen et al. (2019) was used to estimate vegetation composition. It involved measuring the cover of each vascular plant and moss species within a circular frame (diameter 31 cm) located at the center of each plot. We used "The Plant List (2013)" as a reference for species names. To assess the biomass of living moss, we collected samples measuring 100 cm^2 and determined their dry weight. To determine the rate of root production, we utilized root ingrowth cores that were inserted into the peat in October 2013 and collected one year later, following the methodology described by Laiho et al. (2014). The rate of root production was established by means of root ingrowth cores, as described by Laiho et al. (2014), that were inserted in the peat during October 2013 and subsequently retrieved after a year. Detailed descriptions of these methods are provided in **II**.

2.4 Soil biological properties

The study assessed soil biological properties by measuring 72 peat samples, with 24 samples per age class, each measuring 6×6×10 cm and collected from the sample plots. Two methods were used to evaluate the impact of machinery traffic induced disturbance on microbial carbon and community composition: chloroform fumigation–extraction (FE) (Vance et al. 1987; Voroney et al. 2008) and analysis of phospholipid fatty acids (PLFA) (e.g., Pennanen et al. 1999). To assess the biological activity of the peat soil, we conducted laboratory experiments to measure the potential CO2 production rate, as described by Peltoniemi et al. (2015). Additionally, we evaluated the in-situ decomposition rate of cellulose strips that were inserted into the peat within each plot, following the method described by Lähde (1974). The cellulose strips were retrieved from the peat in each sample plot one year after their insertion. A detailed information is given in **II**.

2.5 Greenhouse gas concentrations in the soil and emissions

Specialized samplers, made out of 2-meter long silicon tubes sealed at both ends and connected to a 1-meter long plastic pipe as described in Kammann et al. (2001), were used to measure gas concentrations (CO_2, CH_4, and N_2O) at depths of 5 and 15 cm of the top soil layer. The first sampling round was conducted in July 2013, and subsequently, it was performed on a monthly basis from May to August 2014, covering the growing season. However, due to technical issues, the N_2O data was available only for 2013. In total, 720 gas concentration samples were used for analysis.

To measure CO_2 and CH_4 emissions in situ, we used cylindrical aluminum chambers (31.5 cm in diameter and 30.5 cm in height) described by Alm et al. (2007). These chambers were placed at 72 fixed points within the plots without any prior removal of vegetation. As a result, both autotrophic and heterotrophic respiration were included in the CO_2 emissions. Gas samples were collected at four different time intervals (5, 15, 25, 35 min) after closing the chamber using a 20 ml syringe. These samples were subsequently transferred to 20 ml vacuum tubes that had been pre-flushed. The tubes were stored in a refrigerator before being analyzed using Agilent Technologies 7890A gas chromatograph with Gilson GX-271 liquid handler, following the same procedure as described in Korrensalo et al. (2018). The air temperature inside the chamber and the soil temperature at the surface and at depths of 5, 15, and 30 cm were measured during gas sampling. Throughout the growing season of 2014 (May-September), chamber measurements were performed five times, resulting in the analysis of a total of 1440 gas samples. Soil CO_2 and CH_4 emissions were calculated based on the linear change in gas concentration over time in relation to chamber volume and temperature.

The water table was measured next to the sample plots concurrently with the sampling of greenhouse gas emissions and concentrations in 2014. For a more details, refer to **II**.

2.6 Data analysis

Normality and homogeneity of variables were tested using the Shapiro-Wilk and Bartlett tests, respectively. Linear mixed effects analysis was performed to investigate soil property recovery following disturbance. The nlme package (Pinheiro et al. 2018) of R (Development Core Team 2015) was utilized for linear mixed effects modeling. Mixed effects models for bulk density, field capacity, parameters of van Genuchten-Mualem equation, and living moss biomass were presented in detail in **I**, while mixed models for annual root production, microbial biomass carbon, CO_2 production potential, and rate of cellulose decomposition were presented in **II**. Recovery of bulk density after disturbance was examined in **I** using nonlinear regression analysis.

We utilized multivariate techniques in **II** to explore how machine traffic and time since harvesting affected vegetation composition and PLFA profiles. Due to significant variations in species composition between plots in the vegetation data, we applied Detrended Correspondence Analysis (DCA) to investigate the vegetation's variation and its association with disturbance class (DC) and age class (AC). Additionally, we used Principal Components Analysis (PCA) to identify corresponding patterns within the PLFA data.

2.7 Modeling water retention characteristic

To develop PTFs, we utilized water retention data from **I**, along with two additional data sets from Southern Finland (Fig. 1) that included measurements of WRC profiles and physical properties of peat. The data set obtained from Päivänen (1973) and from Weiss et al. (1998) consisted of WRC profiles for drained and undrained peatlands in the vicinity of Hyytiälä Field Station (61°50'N; 24°20'E). All data sets contained information about the top layer of peat (up to 15 cm depth), such as the type of peat, bulk density (ρ_b, $kg \cdot m^{-3}$), degree of decomposition (using the von Post scale), loss on ignition (%), ash content (%), total porosity (ϕ_f, %), and water retention characteristics ($m^3 \cdot m^{-3}$). For detailed information concerning the sampling and WRC acquisition, refer to **I**, Päivänen (1973), and Weiss et al. (1998). The final data set comprised 347 WRC profiles, with 80% of them used for prediction and the remaining 20% for validation.

To create PTFs, we utilized feed-forward ANN models, which are widely employed for this purpose with mineral soils. These models generally consist of multiple interconnected layers of neurons categorized as input, hidden, and output layers. The first hidden layer of neurons receives initial information from the input layer and perform computations required to transmit it to the next hidden or output layer. We developed the ANNs using Python 3.7.

Within the project two models were developed, namely the pseudo-continuous and point estimation ANNs. These models had a comparable structure, comprising of an input layer, one hidden layer, and an output layer, as illustrated in Fig. 2. To avoid overfitting, cross-validation was utilized. Additionally, permutation importance method was employed using the ELI5 library (Korobov and Lopuhin 2021) to gain insight into the impact of selected estimators on model performance and to unravel the black box nature of the developed models. Paper **III** provides a more detailed description of this process.

In **III**, we evaluated the ANNs by comparing the predicted and observed water contents at specific matric potentials. The assessment was conducted using Python 3.7. In evaluation we used statistical parameters mean absolute error (MAE), root mean squared error (RMSE), and coefficient of determination (R^2).

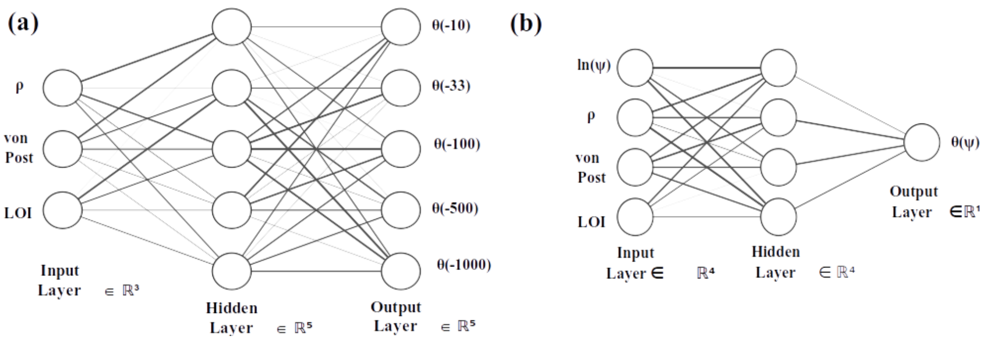


Figure 2. Schematic diagrams of ANN architecture used in the study: (a) point estimation; (b) pseudo-continuous.

3. MAIN RESULTS

Thinning operations caused visible soil deformation in the form of rutting and compaction in the top layer of peat. Analysis of samples and data collected from both undisturbed control points and logging trails revealed that trafficking had negative effects on vegetation and the physical and biological properties of the peat. However, these properties gradually recovered over time. The main change observed following the disturbance was a reduction in the number of macropores, which led to the collapse of the pore structure. This change was reflected in the water retention characteristics, and subsequently impacted the soil environment in the logging trails. Therefore, changes in water retention characteristics could be used to assess the disturbance impact. However, determining these characteristics is a time-consuming and laborious process. To address this, we developed a pedotransfer function based on an artificial neural network, which uses bulk density, von Post index, and total porosity as input parameters to estimate water retention characteristics.

3.1 Soil physical and chemical properties

The analysis of linear mixed effects models revealed that trafficking had led to an increase in peat bulk density. The bulk density in the undisturbed area was 113 $kg \cdot m^{-3}$, which increased to 190 $kg \cdot m^{-3}$ after the trafficking (see Table 3 in **I**). However, over time, the bulk density in DC1 and DC2 gradually recovered towards the intact values (see Table 3 in **I**). The recovery rate, or rate of change, was estimated to be -14.81 $kg \cdot m^{-3}/t$ for DC1 and -13.73 $kg \cdot m^{-3}/t$ for DC2 (see Fig. 2 in **I**).

The water retention characteristics, described by the van Genuchten-Mualem equation (**I**, Eq. 2), were significantly affected by the disturbance class. Specifically, the parameter θ_s, representing the total pore volume, decreased due to trafficking (**I**, Table 2), while the reciprocal of parameter α ($1/\alpha$), which describes air entry potential, increased (**I**, Table 2). Notably, the air entry potential increased to 4.34 kPa and 3.12 kPa in the disturbed plots DC1 and DC2, respectively, while remaining at 0.42 kPa for the control plots. This change is ecologically significant, as air entry potential plays a critical role in soil aeration. Following the trafficking, the total porosity of the disturbed peat decreased by 5-6% (Fig. 3a). However, the most significant change observed was in the pore size distribution, as the volume of meso- and micropores immediately increased reflecting the change in water retention characteristics (Fig. 3b). Although the mean field capacity was higher in all disturbed plots (DC1, DC2), the difference was only significant immediately after the deformation (**I**, Table 1). It is worth noting that the impact of trafficking on water retention and pore distribution gradually recovered over time following the disturbance (Fig. 3).

An upward trend in pH was observed in the ruts of the disturbed peat (**I**, Fig. 5), with the increase occurring immediately after the disturbance and being more pronounced in areas with more severe damage. Nevertheless, we did not observe a significant pH recovery over time in the case of DC2.

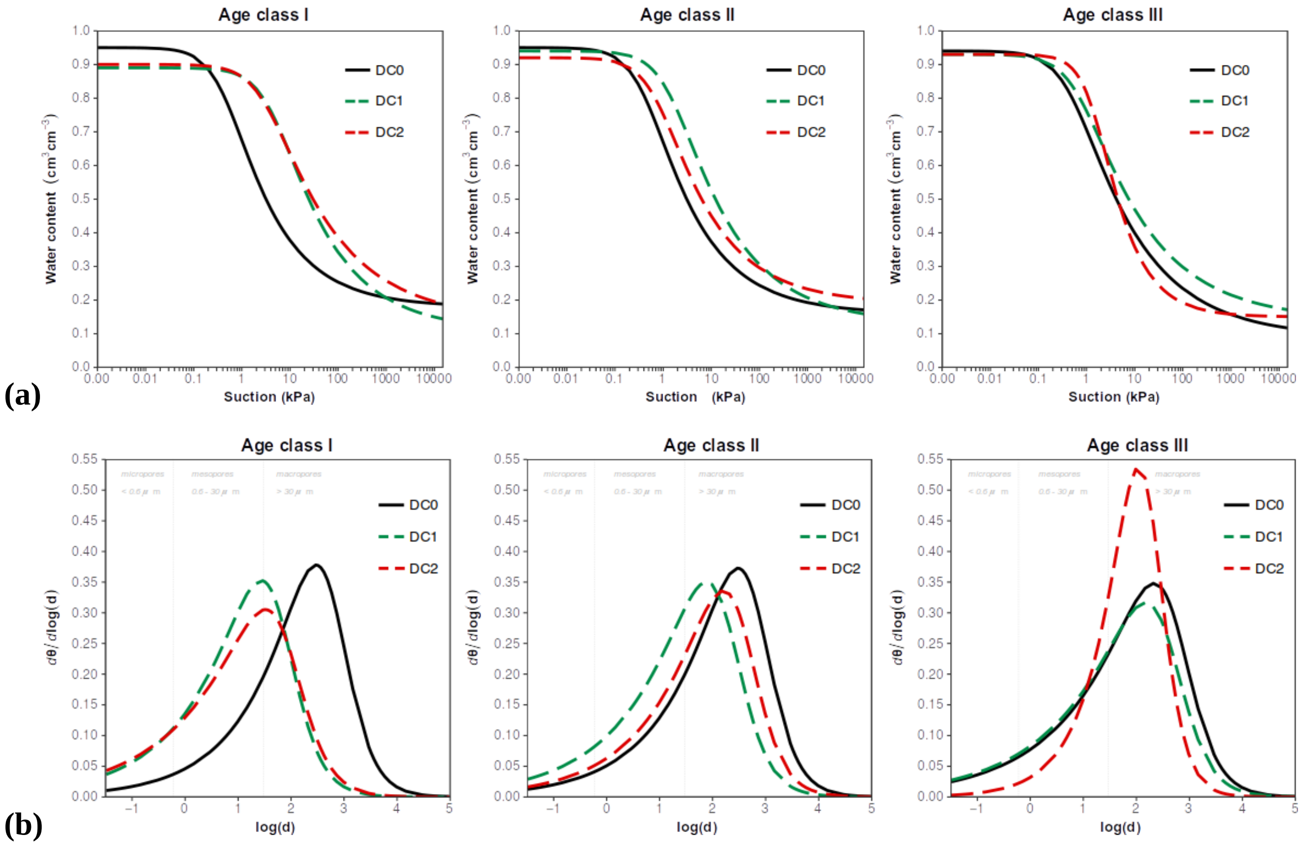


Figure 3. (a) Water retention characteristics in logging trail and undisturbed control points and (b) Pore-size distribution in logging trail and undisturbed control points. Age classes: AC 1: <1 year since thinning; AC2: 4–5 years since thinning; AC3: 14–15 years since thinning. Disturbance classes: DC0: undisturbed; DC1: rut depth < 0.2 m; DC2: rut depth > 0.2 m.

3.2 Vegetation

The vegetation data comprised 23 species, including 9 mosses and 14 vascular plants. The plant species composition in the recently disturbed plots (AC1: DC1, DC2) differed distinctly from the other plots, as can be observed by their separation along DCA Axis 1 (**II**, Fig. 2). Sedges like *Eriophorum vaginatum* and *Carex canescens*, and mosses like *Aulacomnium palustre* and *Sphagnum magellanicum*, were abundant in the disturbed plots of AC1, while peatland and forest dwarf shrubs like *Vaccinium uliginosum* and *V. vitis-idaea* were typical for other plots (**II**, Fig. 2a). There was a change in the species composition in the disturbed plots between AC2 and AC3. Forest herbs such as *Trientalis europaea* and *Dryopteris carthusiana*, and dwarf shrubs such as *Calluna vulgaris* were abundant in the disturbed plots of AC2 in contrast to AC3.

The disturbance led to a decline in living moss biomass, but only in AC1 areas (**II**, Table 3). In contrast, control plots (AC1: DC0) had an average of 608 $g \cdot m^{-2}$ of living moss biomass. In plots with deep ruts (AC1: DC2), living moss was entirely absent, while in plots with shallow ruts (AC1: DC1), moss biomass was reduced to 145 $g \cdot m^{-2}$. However, this effect was not long-lasting, as older sites did not display any significant difference in living moss biomass between disturbed and control plots.

The results of the linear mixed-effects model indicate that root biomass production in the upper 10 cm peat layer was significantly reduced by disturbance (**II**, Table 3), with no notable impact observed at deeper levels. Specifically, in the recently disturbed plots (AC1: DC1, DC2), root production in the upper layer was only 21-36% of the production in undisturbed plots (AC1: DC0). However, in AC2, the root production began to recover, with only the severely disturbed plots (AC2: DC2) demonstrating reduced production (**II**, Table 3). Furthermore, in AC3, there was no significant difference in root production between the control plots (AC3: DC0) and disturbed plots (AC3: DC1, DC2).

3.3 Soil biological properties

The analysis of PLFA profiles using PCA did not show any distinct alterations in the microbial community following the disturbance. Moreover, the impact of traffic did not affect either the microbial biomass derived from PLFA or the C_{mic} determined by FE.

The CO_2 production potential in recently disturbed plots (AC1: DC1, DC2) was twice as high as that in control plots (AC1: DC0) (Fig. 4). In AC2, the CO_2 production potential in moderately disturbed plots (AC2: DC1) was similar to that in control plots (AC2: DC0), while it remained high in severely disturbed plots (AC2: DC2). These results (Fig. 4) suggest that the recovery of CO_2 production potential depends on the degree of disturbance, with a more rapid recovery after moderate disturbance.

The decomposition rate of cellulose was not impacted by the degree of disturbance or age class. The uppermost 5 cm layer of peat had the highest rate of decomposition, averaging around 0.68 $year^{-1}$, which decreased as with increase in depth.

Figure 4. Carbon dioxide (CO_2) production potential per gram of soil dry mass; DC0, DC1, and DC2 denote disturbance classes 0, 1, and 2, respectively. Different lowercase letters in a column indicate significant ($p < 0.05$) differences in interaction effects of age class and disturbance by Tukey pairwise comparison. Age classes: AC 1: <1 year since thinning; AC2: 4–5 years since thinning; AC3: 14–15 years since thinning. Disturbance classes: DC0: undisturbed; DC1: rut depth < 0.2 m; DC2: rut depth > 0.2 m.

3.4 Greenhouse gas concentrations and emissions

According to the results of mixed-effects model (II, Table 4), disturbance led to an increase in soil CO_2 concentrations. The recently disturbed plots (AC1: DC1, DC2) showed significantly higher CO_2 concentrations at depths of 5 cm and 15 cm compared to control plots. Severe disturbance still had an impact on older age classes (AC2 and AC3), with higher CO_2 concentrations observed in DC2. In general, the highest CO_2 concentrations were observed in the deeper peat layer (15 cm).

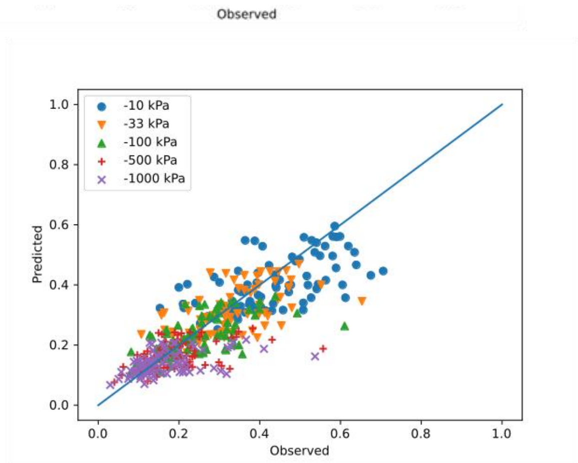
Similar to CO_2 concentrations, CH_4 concentrations in peat exhibited similar patterns (II, Table 4). However, unlike CO_2 concentrations, a significant increase in CH_4 levels in response to disturbance was only evident in the deeper peat layer (15 cm). Moreover, higher mean concentrations were still observed in older DC2 plots. Overall, the deepest peat layer (15 cm) exhibited the highest CH_4 concentrations, similar to CO_2 (**II**, Table 4).

The measurements of N_2O concentrations, taken in October 2013, showed that the impact only in 15 cm peat layer. This effect was similar to CH_4 (**II**, Table 5). In contrast to the carbon gases, N_2O concentrations were greatest in the control plots at a depth of 15 cm (0.35 $\mu l \cdot l^{-1}$; AC1: DC0), and decreased with disturbance to 0.17 $\mu l \cdot l^{-1}$ (AC1: DC1) and 0.08 $\mu l \cdot l^{-1}$ (AC1: DC2).

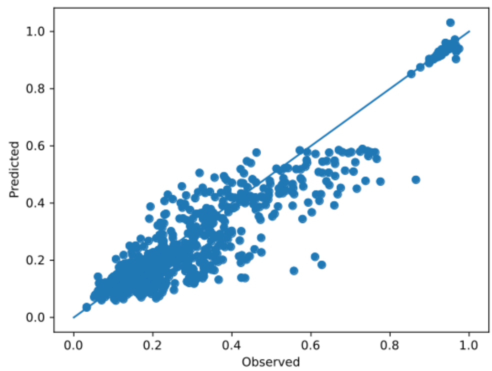
Despite the consistently higher water table levels and moisture content observed in the disturbed plots DC1 and DC2 across all age classes in comparison to the undisturbed control (DC0), the disturbance did not impact the emissions of CO_2 and CH_4, as evidenced by Table 6 in **II**. This stands in contrast to the clear effects exhibited by the soil concentrations. The CO_2 emissions, which encompassed both heterotrophic and autotrophic respiration, exhibited a range of 216 to 18371 $mg \cdot m^{-2} \cdot d^{-1}$. In contrast, the CH_4 fluxes were predominantly low and varied between acting as a source or a sink for the atmosphere, with fluxes ranging from -70 to 186 $mg \cdot m^{-2} \cdot d^{-1}$ (negative values denote uptake).

3.5 Predicting of water retention characteristic

The statistical parameters and scatter plots were used to evaluate the performance of two models in predicting water content at different matric potentials. The results, as presented in Table 2 of **III** and Fig. 5, demonstrated that the pseudo-continuous PTF model performed better than the point estimation PTF model. While both models had comparable RMSE values of 0.079 and 0.077, respectively, the determination coefficient (R^2) for the pseudo-continuous PTF model was higher at 0.92, indicating a better capture of the variance. Conversely, the point estimation PTF model had an R^2 of only 0.37. The permutation importance analysis (**III**, Table 3) of the pseudo-continuous PTF model revealed that the technical estimator was the most important factor, followed by bulk density and LOI.



(a)



(b)

Figure 5. Scatter plots of observed versus predicted water contents (cm^3 cm^{-3}) for **(a)** point estimation PTF at -10 kPa, -33 kPa, -100 kPa, -500 kPa, and -1000 kPa and **(b)** pseudo-continuous PTF at all matric potentials

4. DISCUSSION

4.1 Impact of harvesting machines on peat and vegetation

Mineral soils with fine textures are known to be vulnerable to deformation caused by heavy forest machinery, which can lead to a significant increase in bulk density. This increase can result in changes to the soil's total porosity, pore-size distribution, and connectivity, as well as its water retention and other related properties. However, studies on how peat soils respond to heavy machinery are limited, despite the likelihood of similar changes occurring. It is worth noting that mineral soils rich in organic matter are highly susceptible to compaction (Horn et al. 2007). Our research demonstrated that the bulk density of ruts created by heavy machinery on peat soils increased by 76%, which is significantly higher than the increase typically observed in mineral soils (e.g., Frey et al. 2009). This finding is consistent with a previous laboratory study that observed a 79% increase in bulk density in Sphagnum peat samples under surface pressures comparable to those exerted by heavy machinery (Chow et al. 1992).

Soil water retention characteristics are a crucial physical property that defines the soil's ability to retain water against gravity. While bulk density is related to it, water retention characteristics primarily depend on the soil's internal structure. Therefore, it is a more appropriate descriptor of changes caused by machinery impact. Pore structure determines soil aeration, water regime, and biogeochemical processes, making water retention characteristics an essential parameter to consider. To assess the impact of soil deformation resulting from trafficking, we analyzed changes in water retention characteristics. Previous studies (Hansson et al. 2018; Startsev and McNabb 2001) have demonstrated that mineral soils experience a decrease in macropores and an increase in micropores, leading to higher air entry potential and field capacity. Macropores in its turn play a critical role in soil drainage (Ampoorter et al. 2007). Similarly, Chow et al. (1992) observed a decline in total porosity and volume of larger pores in compacted *Sphagnum* peat. Our study found an increase in field capacity due to trafficking. However, this increase is accompanied by a shift in pore size distribution towards smaller pores, which reduces the saturated hydraulic conductivity (van Genuchten 1980). Consequently, it results in wetter conditions and lower oxygen supply in the soil, in conjunction with an increase in air-entry potential and field capacity.

Our study revealed fascinating insights into the profound impact of soil deformation. Not only did it cause physical changes such as alterations in pore size distribution and bulk density, but it also affected physical properties that control water retention and triggered chemical changes in the peat. A noteworthy finding was the trend of increased pH values after deformation, which we attribute to limited gas exchange that resulted in heightened concentrations of CO_2 and CH_4. This phenomenon occurs when waterlogging limits O_2 diffusion into the soil, leading to the emergence of anaerobic conditions that shift microbial activity toward alternative electron acceptors. Consequently, there is a reduction in N, Mn, Fe, and S during anaerobic respiration, and CO_2 by methanogenesis. The release of electrons induces electron activity (pe) + pH drops causing reduction and leading to higher pH values.

Mechanical disturbances can bring about changes in the structure of vegetation, and these changes can be detected through differences in plant community composition between disturbed (AC1: DC1, DC2) and undisturbed (AC1: DC0) plots in the AC1 sites. Sedges, including *Eriophorum vaginatum* and *Carex canescens*, which are known to thrive after disturbance events like clearcutting and restoration (Komulainen et al. 1999), had significantly higher coverage in the recently disturbed plots (AC1: DC1, DC2). This observation is similar to that of Davidson et al. (2021), who reported a shift in vegetation community of seismic lines towards sedge dominance. The changes in vegetation composition are likely due to the increased availability of nutrients after harvesting, caused by the mechanical crushing of fresh organic matter and peat aggregates during deformation. The removal of tree biomass can also reduce competition with ground-layer plant species. However, the subsequent recovery of forest herbs and dwarf shrubs is probably driven by the closure of the canopy, which has a negative impact on several wetland species (Kokkonen et al. 2019).

The living moss biomass was greatly disturbed due to the mechanical removal of moss by wheels/tracks during harvesting, resulting in an immediate reduction in the recently disturbed plots (AC1: DC1, DC2). Nevertheless, our study found that there was a rapid recovery in both moss biomass and *Sphagnum* moss cover. This finding is consistent with previous studies that have shown a quick response by mosses to disturbance, both in terms of sensitivity and recovery, as reported by Hannerz and Hånell (1997), Deans et al. (2003), Zhu et al. (2019), and Deane et al. (2020).

The reduction in root production observed in the upper peat layer (10 cm) of the recently disturbed plots (AC1: DC1, DC2) is likely linked to the removal of tree biomass and subsequent decrease in fine root production. Additionally, our study noted reduced soil aeration and macropore size in the freshly formed ruts, which may lead to a shift towards anaerobic processes (Frey et al. 2009) and create a hostile environment that inhibits fine root growth, similar to mineral soils with high clay content and impermeable layers (Rhoades et al. 2003). However, we observed a recovery in root production in our study sites, particularly in the deeper peat layer, which may be attributed to an increase in sedges that are capable of tolerating anoxic soil conditions.

The CO_2 concentrations recorded in the ruts of the recently disturbed plots (AC1: DC1, DC2) at depths of 15 cm, ranging from 2-6%, were similar to those reported in other studies conducted under forest vegetation (Allman et al. 2016; Jankovský et al. 2019; Magagnotti et al. 2012; Neruda et al. 2010). Neruda et al. (2010) identified a concentration of 0.6% CO_2 in soil air as a threshold value indicative of significant changes in soil structure and the potential impact on root growth. Meanwhile, Erler and Güldner (2002) suggested that CO_2 concentrations exceeding 2% can completely hinder biological recovery. In our study, the CO_2 concentration in the soil in all recently disturbed plots (AC1: DC1, DC2) exceeded this value by several folds, which could have resulted from machine traffic-induced structural changes in the soil. We also observed greater spatial variability in CO_2 concentrations in the severely disturbed plot (DC2), likely associated with changes in soil structure caused by machine traffic. However, the lower CO_2 levels measured in older sites generally indicate that the soil has recovered over the 15-year period.

Disturbance resulted in a rise in the CO_2 production potential of the peat soil, which was consistent with the higher CO_2 concentrations in the same plots. CO_2 production in soil is typically influenced by both root respiration and decomposition of organic matter (Ball et al. 1999). However, since the CO_2 potential was measured in the laboratory from peat without roots, and root production was reduced due to disturbance, it suggests that the increase in CO_2 production potential was mainly driven by an accelerated rate of organic matter decomposition. The ruts created during disturbance likely brought fresh peat from deeper layers to the surface, while mechanical milling by forest machinery provided decomposers with crushed organic matter particles. This increase in decomposition was likely due to these factors, and the mechanical milling appears to have played a more significant role as the decomposition of cellulose was largely unaffected.

Despite observing increased CO_2 production potential and soil CO_2 concentrations in the ruts, there was no significant difference in net soil CO_2 emissions between the disturbed and non-disturbed control areas, which was unexpected. This lack of response is in line with the absence of a clear difference in microbial biomass and community structure between the two areas. Previous studies have also shown contradictory responses of soil CO_2 emissions to disturbance (Novara et al. 2012; Pearson et al. 2012). Although it is challenging to explain why soil CO_2 and CH_4 concentrations increased while emissions remained unaffected, our findings align with a previous study that demonstrated a decoupling between CO_2 emissions and production in the soil layers (Barry et al. 2020).

Earlier studies have assessed the soil C balance in drained peatland forests (e.g., Ojanen et al. 2013). These forests may roughly be divided into two classes based on their soil C sink/source behavior: nutrient-poor sites, in which the soils may be either small sources or sinks of C, and nutrient-rich sites, where the soils are constantly losing C. While drained and managed peatlands generally are GHG-emission hotspots, boreal nutrient-poor drained peatland forests, such as our study sites, are the only drained peatlands where this may not be the case (Olsson et al. 2019). Our GHG emission measurements were not designed for evaluating soil C balance; however, based on them we may conclude that the thinning operations as such do not seem likely to cause notable site deterioration in that respect. Our study could not be extended to cover further the nutrient-rich forests that have different vegetation and soil characteristics (Päivänen and Hånell 2012), and the results may thus not necessarily be generalized to those.

In contrast to CO_2, CH_4 concentrations were only significantly higher at a depth of 15 cm in the recently disturbed plots (AC1: DC1, DC2). The formation of ruts during disturbance changed the site's microtopography, resulting in water-induced anaerobic conditions with a higher water table. The relatively quick recovery may be attributed to the depletion of readily available substrates for methanogenesis as the substrates produced from mechanical milling were quickly utilized. This aligns with our observation of a higher water table in disturbed plots across different age classes. We also found that harvesting has altered soil bulk density, increased water retention by reducing mesopore volume, which is common in mineral soils after machinery impact (Cambi et al. 2015; Frey et al. 2009; Grigorev et al. 2021; Magagnotti et al. 2012), leading to increased anaerobic conditions. However, we did not find any significant difference in the net emissions of CH_4 between the ruts and the adjacent non-impacted control areas. This contrasts with the findings of Strack et al. (2018), who reported that increased bulk density values associated with track formation and increased graminoid cover led to increased CH_4 emissions from Canadian peatland sites. It is noteworthy that in our study, despite the increased bulk density and sedge cover, as well as an increased water table level and soil CH_4 concentration, we did not observe the same pattern. This suggests that CH_4 emissions in logging trails more influenced by high oxidation than low production. As a result, the potential for CH4 emissions is considerable in logging trails within drained peatland forests, especially in the immediate aftermath of disturbance.

4.2 Recovery from disturbance

The recovery of mineral soil following deformation depends on various factors such as soil type, conditions, and the extent of impact, and in some cases, the damage may be irreversible (Cambi et al. 2015; Horn et al. 2007). However, our study on peat soil reveals a high level of resilience and a relatively fast recovery following thinning operations. The impacted peat recovered within our chronosequence’s timeframe, with an initial rapid recovery in bulk density that gradually slowed. Root ingrowth, wind-induced root movements, freezing-thawing cycles, and moisture-induced shrinking-swelling cycles in peat are some factors that support recovery (Chang 2013; Finér and Laine 1998). During the recovery, the bulk density range was similar to that of peat under a winter road that had been closed for four years (160 $kg \cdot m^{-3}$) (Strack et al. 2018), while the bulk density in undisturbed areas varied from 40 to 110 $kg \cdot m^{-3}$. The water retention characteristics showed a similar recovery pattern to that of the bulk density.

The peat soil’s ability to recover relatively quickly suggests that it could potentially endure frequent harvesting cycles that could accompany a transition to continuous-cover forestry. This practice has been proposed as a means to alleviate the negative environmental impacts of traditional rotation-based forestry on peatlands (Nieminen et al. 2018). In contrast, mineral soils’ slow recovery rates (Magagnotti et al. 2012) limit the frequency of logging associated with continuous-cover forestry. A 15-year harvest interval has been recommended to achieve optimal economic returns for spruce-dominated peatland stands under continuous-cover forestry (Juutinen et al. 2020). However, the practical implementation of continuous-cover forestry remains uncertain, and further investigation may be necessary to examine the soil’s recovery following repeated harvest cycles. Moreover, it’s important to note that clearcutting causes more significant soil disturbance than the thinning operations studied here, which may result in a longer recovery period.

4.3 Water retention characteristic

In our study, we identified water retention characteristics (WRC) as a crucial indicator of soil condition after harvesting procedures and a useful monitoring tool. The results of our study demonstrated that the pseudo-continuous model, a neural network trained with fitted data, can accurately predict WRC for boreal peatlands. Unlike mineral soils, the reliable predictors for WRC in peat are bulk density, degree of decomposition (von Post scale), and LOI. Additionally, the pseudo-continuous PTF outperformed the point estimation PTF, as it can predict water content at any matric potential, making it a more universal model. The pseudo-continuous approach also enables the use of multiple data sets containing water contents at different matric potentials, which is particularly useful for developing PTFs with limited available data. Consequently, the PTF developed in this study can be employed to estimate WRC for boreal peatlands. Further, the ANN-based proxy determination of peat water retention capacities could have a promising perspective for further analysis of hydrological regime regulations related to long-term carbon sequestration in managed peatlands.

5. CONCLUSION

This study established a link between the changes in peat structure, physical properties, physical conditions, and the soil biology and vegetation of drained peatlands following forest thinning operations. This finding is significant because it provides a foundation for physics-based modeling of water and gas exchange in the logging trail environment and enables the assessment of the ecosystem's recovery from the impact. In addition, the water retention characteristics presented in the study facilitate our understanding of the drained peatland ecosystem's impacts and responses to forest thinning operations.

The results of this study have significant implications for evaluating **(a)** the impact of forest machinery traffic on peat soil, **(b)** sustainability of peatland forestry, and **(c)** the threshold of soil disturbance beyond which restorative actions should be taken to reduce physical and biological damage. The study shows that thinning operations do not cause permanent changes to peat soil properties, suggesting that these soils may be resilient to the disturbance caused by forest machinery. It is worth noting that the study solely examines initial thinnings, implying that frequent or severe disturbances may have more significant long-term consequences on soil properties.

While this study examines the impact of harvesting on previously drained peatland used for forestry, it is important to note that in the viewpoint of the EU Nature Restoration law peatland forestry may not be the best option as it causes a risk to the most space-effective carbon store, which pristine peatlands represent (Jurasinski et al. 2023). Thus, there is pressure to restore these forested peatlands to their natural state. As the restoration of forested peatlands includes harvesting of the tree stand formed after the drainage my results on the recovery of logging trails are promising also for the restoration success. This, however, would need further investigation.

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