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Effects of Drainage on Carbon Stock in Hemiboreal Forests: Insights from a 54-Year Study

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Abstract: In the Northern Hemisphere, forests play an important role in carbon storage. During the past few decades in the eastern Baltic and Nordic regions, forest drainage has been a common occurrence, which also has an effect on carbon stock. Most of the studies on this issue were carried out in boreal zones and were focused on short-term effects. Thus, our aim was to evaluate the long-term (after 54 years) effect of drainage on carbon stock (CS) changes in organic soil (Fibric histosols) in hemiboreal forests. Three forest types were selected in drained (*Myrtillosa* turf. mel (Mmel)) and undrained (*Caricoso–phragmitosa* (CP) and *Sphagnosa* (Sph)) parts of the same area. Surface level changes, soil penetration resistance, and soil and tree biomass carbon stock were assessed to evaluate the drainage effect. Drainage caused an average surface level drop of 25 cm, but did not deplete the soil carbon pool, resulting in significantly and substantially higher (2 to 6 times) tree biomass carbon stock. The drainage of organic soils in managed wet forests leads to an increased long-term contribution to climate change mitigation, thus such areas should be established or maintained in conjunction with areas that maximize other ecosystem services to ensure the sustainability of forest landscapes.

Keywords: carbon storage; climate change mitigation; drained forest; organic soil

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1. Introduction

Boreal forests play an important role in global carbon stocks by providing approximately one-third of terrestrial carbon storage [1]. Many peat-forming ecosystems in the eastern Baltic and Nordic regions have been converted into forest land to increase tree biomass production, mostly by improving the growing conditions by reducing excess water using drainage systems consisting of ditches [2,3]. In vast areas of the boreal zone, a high water content in the soil hampers tree growth, especially at sites with mineral or peat soils [4,5]. This is because gas exchange occurs much faster in aerated soils than in water-saturated soils [6]. Furthermore, the subsequent limited oxygen availability due to high groundwater levels negatively affects the functioning of plant roots in terms of oxygen uptake. Therefore, a consistently high groundwater level results in shallow rooting, which leads to higher mortality and impairs tree growth [7].

In the boreal biome, forest soils hold more carbon than overstories [8–11]. Organic soils can be sources of both the emission and removal of greenhouse gases (GHGs; nitrogen (N) and carbon (C)) [12]. Organic soils are produced from partially decomposed plant material under anaerobic conditions through a gradual process of accumulation and compression in peat-forming ecosystems, and they are typically below the high water table [2]. It is known that in organic soils, drainage leads to enhanced aerobic decomposition [13,14], so the mobilization of C and N stores can occur. Previous studies have shown that drainage systems can have both positive and negative impacts on GHG emission and removal at the landscape level [15–17]. Proper drainage can improve soil aeration, which is essential for the growth of vegetation, including trees [18]. This can lead to increased carbon sequestration, as trees absorb carbon dioxide from the atmosphere and store it in their biomass [12,19,20]. On the other hand, the establishment and maintenance of drainage systems can also cause soil disturbances, which can alter GHG emissions and removal. For example, the excavation of drainage ditches can release previously stored carbon in the soil, leading to increased emissions of carbon dioxide and other GHGs [21,22]. Additionally, the removal of vegetation and the altering of soil structure during the creation of drainage systems can reduce the overall carbon sequestration potential of the landscape and reduce soil carbon input in the first few years. However, in boreal zones, forestry practices based on ditching can be considered environmentally friendly for nutrient-poor boreal peatlands as the peat soil continues to be a CO_2 sink even after drainage [12]. This may be related to changes in litter quality and increased litter production [23,24].

Recently, in hemiboreal regions there have been studies evaluating different carbon pools of forests on dry, wet, and drained mineral and organic soils [25–27]. According to the National Forest Inventory (NFI), 29.7% of the total forest area in Latvia is located on drained soils. Therefore, it is important to consider the potential impact of drainage systems on soil carbon stock and GHG emissions and removal in order to make informed decisions about how to establish and maintain them. This can help to minimize the negative impact and maximize the positive impact on the environment. Recent studies have focused on assessing soil organic layer thickness and soil organic carbon stock [28], and the carbon budget of drained and undrained organic soils [29]. Moreover, there has been an additional focus on GHG emissions from organic soils and even drainage ditches [30,31]. However, knowledge about the long-term impact of drainage on soil subsidence and the influence of drainage ditches on changes in organic soil carbon stock is lacking; therefore, it is important to assess whether drainage (including the maintenance of amelioration systems) is suitable in terms of carbon storage and climate change mitigation measures. Thus, the aim of this study was to evaluate long-term C storage in organic soils and living tree biomass after drainage. The study hypotheses were as follows: (I) soil carbon stock will be higher at drained sites; (II) tree carbon stock will be higher at drained sites.

2. Materials and Methods

2.1. Location of Sample Plots

This study was conducted in hemiboreal forests in the central part of Latvia (N 56°; E 26°) in a catchment area of the Veseta River, which is dominated by Scots pine (*Pinus sylvestris* L.) and Norway spruce (*Picea abies* (L.) H. Karst.), with an average growing stock of 50 m^3 ha^{-1}. Stand age ranged from 40 to 110 years. The study area, which contains organic soils (Fibric histosols), originally existed as a transitional mire, corresponding to a hemiboreal vegetation zone. A drainage system was established in part of the study area in 1960, which led to changes in dominant forest types. In 1963, a forest research station was established, and three types of sites were established for our study: *Myrtillosa* turf. mel (Mmel, n = 20; representing poorly acidic peat soil formed by drainage of transitional bog), *Caricoso–phragmitosa* (CP, n = 4; representing moderate fertile, wet, and moderately decomposed peat soil), and *Sphagnosa* (Sph, n = 6; representing acidic peat soil, where groundwater is very shallow and peat is poorly aerated due to the high humidity) [32]. The peat depth in the studied sites was 4–4.5 m. The water table level varied among the considered sites (Figure 1).

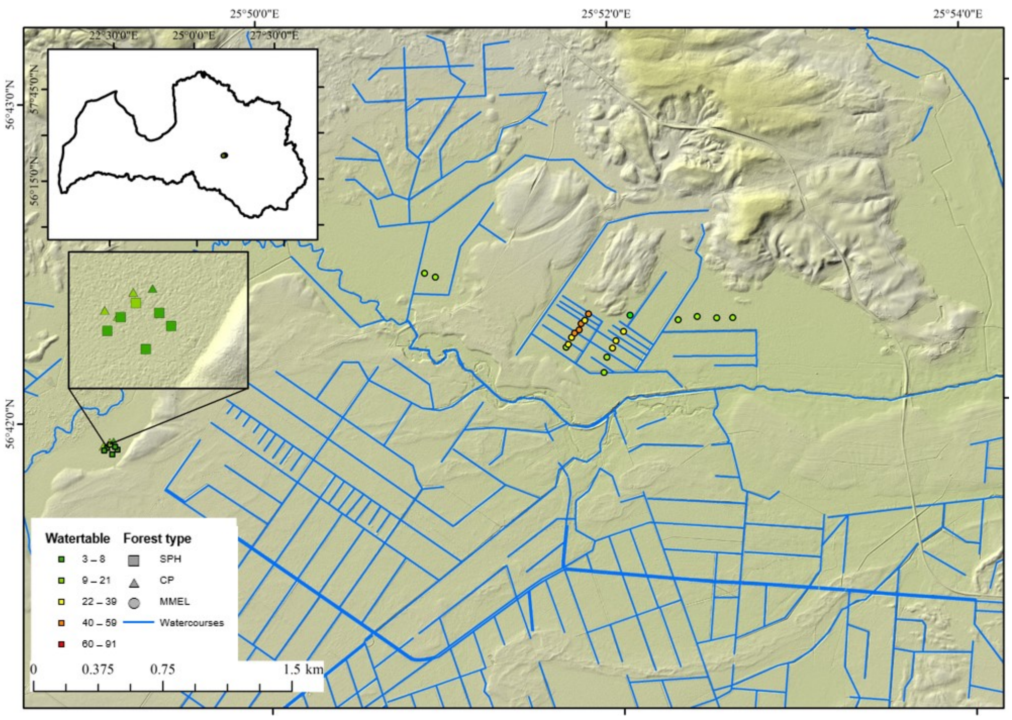

Figure 1. Location of sample plots and level of water table. Colors indicate water table depth; symbols indicate forest types, and the blue lines represent the Veseta River and drainage ditches.

2.2. Field Survey

We established circular sample plots 20 m from transects where the drainage ditches were located. The area of the sample plots was 500 m^2, with a radius of 12.62 m. In each sample plot, tree height and diameter at breast height (DBH) was measured during the last ground surface elevation measurement campaign for all trees > 14 cm to assess stand taxation indices and tree carbon stock. To determine morphometric stand characteristics, we established subplots with a radius of 6.64 m inside the circular sample plots, starting from the same center of the plots. The subplots were divided into quarters. In the quarters located between the north and east cardinal directions (0–90°), DBH was measured for all trees with a diameter of 2 to 6 cm. In the subplots, we determined the species, DBH, and height of the trees. In the 500 m^2 sample plots, we measured the stem diameter at both ends for all lying and standing deadwood trees > 6 cm.

Ground surface elevation was measured using an SEOP DS32 (Tianjin, China) optical nilevel instrument. Ground surface elevation was re-measured several times at the same measurement points in 1960, 1966, 1970, 1972, 1975, and 2014. All measurements in every data survey were carried out at the same drainage ditches. The number of sample plots was equal to the number of measurements of ground surface elevation. P.Zālītis, a scientist who participated in the installation of the drainage systems, took part in the field survey as a consultant.

Resistance to soil penetration was determined at three points in each sample plot: in the center of the plots, 7 m to the east from the center of the plots, and 7 m to the west from the center of the plots. A Royal Eijkelkamp penetrologger (Giesbeek, The Netherlands) was used to determine the resistance to penetration of the soil at depths up to 80 cm. Soil samples were collected in three replicates at each sample plot at four depths: 0–10 cm, 10–20 cm, 20–40 cm, and 40–80 cm. The volume of soil samples was 100 m^3. Sampling was performed using non-disturbing probes by digging soil pits and taking composite samples from a side of each pit at the top, middle, and bottom of each layer. Litter samples were collected in 10 × 10 × 10 cm boxes in three replicates (at the same location where the soil samples were collected).

2.3. Sample Preparation and Measurements

To determine the dry bulk density (kg m^{-3}) and carbon content (g C kg^{-1}) values of the soil samples, the soil and litter samples were dried at 105 °C until they reached constant mass (at least 48 h). A change in mass of less than 0.01% over 4 h was considered constant. The samples were cooled in a desiccator and weighed immediately after being removed. After drying, the soil samples were sieved and ground using an IKA A 11 basic analytical mill. Soil carbon content was determined using a LECO CR analyzer at a temperature higher than 900 °C. Peat subsidence was considered while calculating carbon stock (CS) changes. CS at the control sites was equal to soil CS at 0–80 cm depth, but at the drained sites, it was equal to soil CS at 0–(80–x), where x is subsidence (cm). Changes in CS were calculated as the difference between the CS profiles at the drained and control sites. All fine fallen deadwood samples were stored in a refrigerator using airtight packaging. The samples were dried at 105 °C until they reached constant mass for at least 3 days.

2.4. Calculations and Data Analyses

To assess the differences between stand parameters, we calculated the total biomass, total density, basal area, mean diameter, and height of the dominant species. The volume of living trees and of standing deadwood trees was calculated according to Liepa (Equation (1)) using DBH and tree height values with the respective coefficients for the species [33]:

$$v = \psi \times L^{\alpha} \times DBH^{\beta \lg L + \varphi}, \tag{1}$$

where v is stem volume (m^3); L is stem length (m); DBH is diameter at breast height (cm); ψ α β φ are species-specific coefficients, according to Liepa [33].

The volume of snags and of lying deadwood was calculated according to the formula for a cylinder multiplied by the basic density of a specific tree species in relation to its decay class [34]. The living tree biomass (above- and belowground) was estimated from the DBH and tree height for individual trees (coefficients for specific tree species are summarized in Table 1) based on the local biomass equation (Equation (2)) [35]:

$$lnY_{ki} = \ln(a) + b \times \ln(DBH) + u_k + \varepsilon_{ki}, \tag{2}$$

where Y is the predicted dry biomass of each component of tree i in stand k (kg); H is tree height (m); DBH is diameter at breast height (cm); a, b are regression coefficients (Table 1); u is the random effect for stand k; and ε is the random effect for tree i in stand k.

Table 1. Coefficients of biomass equations for Scots pine and Norway spruce according to Liepiņš [35].

Coefficient	Scots Pine		Norway Spruce	
	a	**b**	**a**	**b**
Stem	2.94	0.020	2.82	0.040
Living branches	1.576	0.201	1.639	0.336
Dead branches	2.102	0.006	3.289	0.000
Stump	2.442	0.007	2.698	0.004
Coarse roots	3.227	0.001	2.998	0.004
Fine roots	1.82	0.017	1.843	0.026

The CS in the living tree biomass was calculated based on the living tree biomass values multiplied by a carbon content of 50% [36,37]. The soil CS was calculated using the carbon content and sample density indicators.

To evaluate the relationship between penetration resistance, soil depth, and water table level, the linear mixed effect model was implemented using the lme4 package. To evaluate the relationship of soil carbon stock between the analyzed site and soil depth, the linear mixed effect model was used, where the significance of the tested fixed and random effects was evaluated by the maximum likelihood approach and the χ^2 criterion.

Tukey's HSD test was used to compare the significant levels of factors. All data analyses were carried out in the program R version 4.0.3 [38] using the lme4, dplR, and emmeans packages [39–41].

3. Results

3.1. Stand Parameters

The main forest inventory parameters (mean DBH, mean tree height, mean basal area, and mean volume) were determined during the study. The mean (mean ± SD) DBH was 20.75 ± 3.51 cm at the Mmel sites, 14.70 ± 3.06 cm at the CP sites, and 7.90 ± 2.33 cm at the Sph sites, and the differences between the sites were statistically significant in all cases ($p < 0.01$). The mean dominant tree height (mean ± SD) was 18.58 ± 2.01 m at the Mmel sites, 15.52 ± 2.07 m at the CP sites, and 9.48 ± 2.75 m at the Sph sites, and the differences between the sites were statistically significant in all cases ($p < 0.01$). The mean (mean ± SD) volume of dominant species was 197.47 ± 70.04 m^{-3} ha at the Mmel sites, 90.72 ± 42.07 m^{-3} ha at the CP sites, and 32.47 ± 35.04 m^{-3} ha in the Sph sites, indicating a positive impact of forest drainage on tree growth (height, DBH, basal area, and volume) because statistically significant differences ($p < 0.01$) were observed only between the drained sites (Mmel) and wet sites (CP and Sph). The mean (mean ± SD) basal area was 20.95 ± 6.84 m^2 ha^{-1} at the Mmel sites, 11.06 ± 4.48 m^2 ha^{-1} at the CP sites, and 4.78 ± 4.75 m^2 ha^{-1} at the Sph sites, and statistically significant differences ($p < 0.01$) were observed only between the drained and wet sites; thus, the wet sites did not differ significantly.

3.2. Soil Parameters

Based on the linear mixed effect model, forest site, soil depth, and water table level showed a complex effect on soil penetration resistance (Tables S1 and S2). At the Mmel sites, soil penetration resistance for all soil depths was similar ($p > 0.05$). At a soil depth of 0–10 cm, the mean (mean ± SD) penetration resistance was 0.47 ± 0.02 MPa. At 10–20 cm, it was 0.43 ± 0.02 MPa; at 20–40 cm, it was 0.43 ± 0.02 MPa, and at 40–80 cm, it was 0.45 ± 0.02 MPa.

3.3. Soil Characteristics

At the CP sites, soil penetration resistance was significantly lower ($p < 0.01$) at a soil depth of 40–80 cm compared to soil depths of 0–10 and 10–20 cm. At a soil depth of 0–10 cm, the mean (mean ± SD) penetration resistance was 0.76 ± 0.05 MPa. At 10–20 cm, it was 0.82 ± 0.05 MPa; at 20–40 cm, it was 0.64 ± 0.05 MPa, and at 40–80 cm, it was 0.49 ± 0.05 MPa.

At the Sph sites, the highest soil penetration resistance was found at a depth of 10–20 cm, which was significantly higher ($p < 0.01$) than at depths of 20–40 and 40–80 cm. At soil depths of 0–10 cm, the mean (mean ± SD) penetration resistance was 0.76 ± 0.04 MPa. At 10–20 cm, it was 0.89 ± 0.04 MPa; at 20–40 cm, it was 0.61 ± 0.04 MPa, and at 40–80 cm, it was 0.47 ± 0.04 MPa. At soil depths of 0–10 cm and 10–20 cm, soil penetration resistance was significantly higher ($p < 0.01$) at the Sph and CP sites than at the Mmel sites, but the values were similar between Sph and CP cm. All significant differences between the soil depths of the soil and the analyzed sites are shown in Figure 2.

Soil subsidence was elevated only at the Mmel sites. Our results show that soil subsidence occurred after drainage (Figure 3). The most intensive soil subsidence occurred during the first years after the amelioration system was established; 10 years after drainage, the top layer of the soil dropped by 12.3 ± 1.9 cm, and after 15 years, the top layer dropped by 15.8 ± 2.1 cm. Peat subsidence decreased over the next 40 years. From 1974 to 2014, the top layer of the soil dropped by 9.9 cm to 25.7 ± 3.5 cm from the initial level (Figure 3).

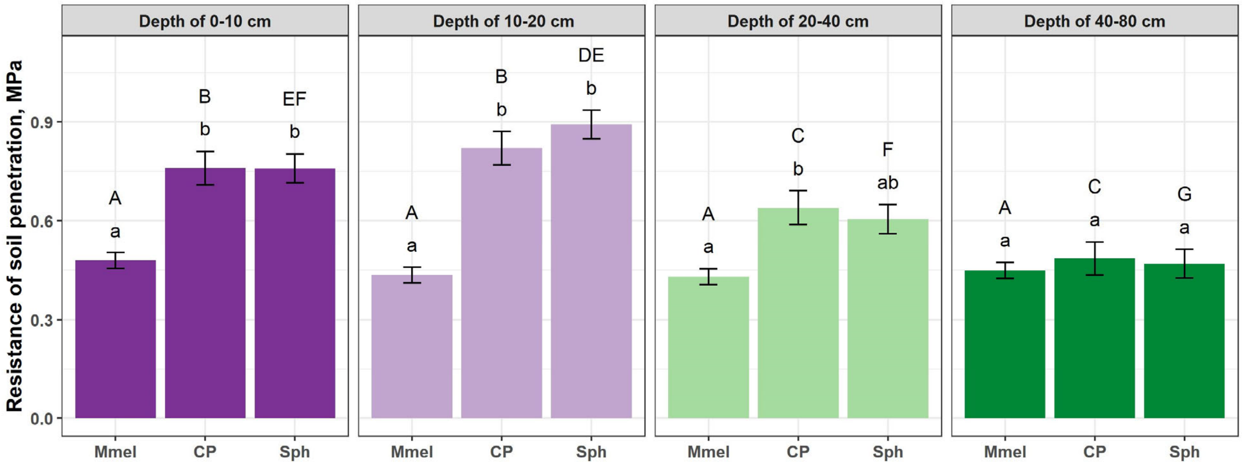

Figure 2. Resistance to soil penetration between certain depths of soil. Mmel, *Myrtillosa* turf. mel. sites; CP, *Caricoso–phragmitosa* sites; Sph, *Sphagnosa sites*. Different letters above bars represent significant differences ($p < 0.05$) in estimated means, as determined using Tukey's HSD test. Lowercase letters indicate significant differences between forest sites within each depth; uppercase letters indicate significant differences between soil depths within forest sites.

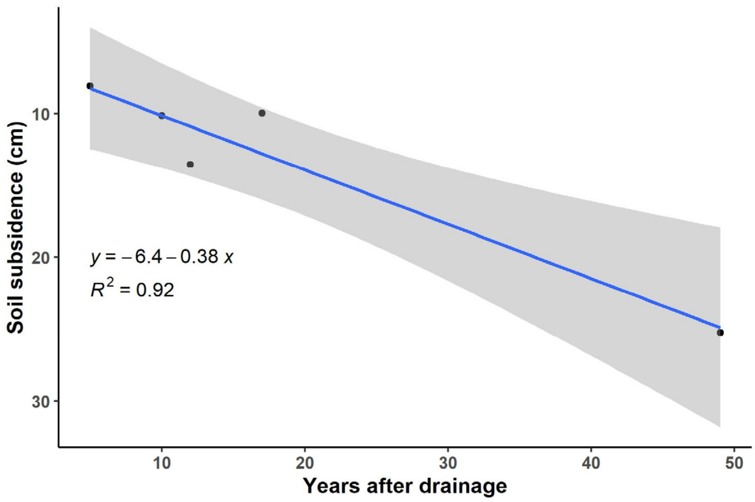

Figure 3. Differences in soil subsidence at various times after forest drainage at *Myrtillosa* turf. mel. (Mmel) sites.

3.4. Carbon Stock

Carbon stock changes in the living tree biomass were evaluated based on the carbon stock in the stems, living branches, dead branches, coarse roots, fine roots, and stumps (Table 2). Soil carbon stock was calculated based on all analyzed soil depths (Table 2). Overall, carbon stock in living tree biomass at the Mmel sites was significantly higher (p value < 0.01) than at the CP and Sph sites (Figure 4).

Table 2. Carbon stock in biomass and soil (mean ± SD). Mmel, *Myrtillosa* turf. mel. sites; CP, *Caricoso–phragmitosa* sites; Sph, *Sphagnosa* sites. O horizon in soil refers to organic layer at soil surface.

Carbon Stock (Tons ha^{-1})	Mmel	CP	Sph
Stem	53.54 ± 15.14	20.94 ± 10.95	6.08 ± 6.94
Living branches	16.20 ± 8.11	7.93 ± 3.21	6.28 ± 5.10
Dead branches	1.38 ± 0.43	0.91 ± 0.34	0.37 ± 0.55
Coarse roots	11.86 ± 4.76	3.55 ± 1.32	1.40 ± 1.67
Fine roots	12.04 ± 4.83	3.61 ± 1.34	1.42 ± 1.70

Table 2. *Cont.*

Carbon Stock (Tons ha^{-1})	Mmel	CP	Sph
Stump	2.47 ± 1.04	1.31 ± 0.51	0.85 ± 0.78
Total in biomass	97.48 ± 27.41	38.27 ± 14.67	16.39 ± 16.12
O horizon	5.08 ± 2.06	0	0
Soil depth 0–10	74.40 ± 16.34	41.80 ± 7.30	37.95 ± 4.52
Soil depth 10–20	67.45 ± 15.01	42.12 ± 8.96	36.10 ± 5.29
Soil depth 20–40	126.55 ± 28.33	85.87 ± 18.83	75.97 ± 16.65
Soil depth 40–80	239.73 ± 52.12	208.12 ± 46.22	189.25 ± 40.31
Total in soil	513.21 ± 106.26	377.92 ± 79.51	339.28 ± 63.77

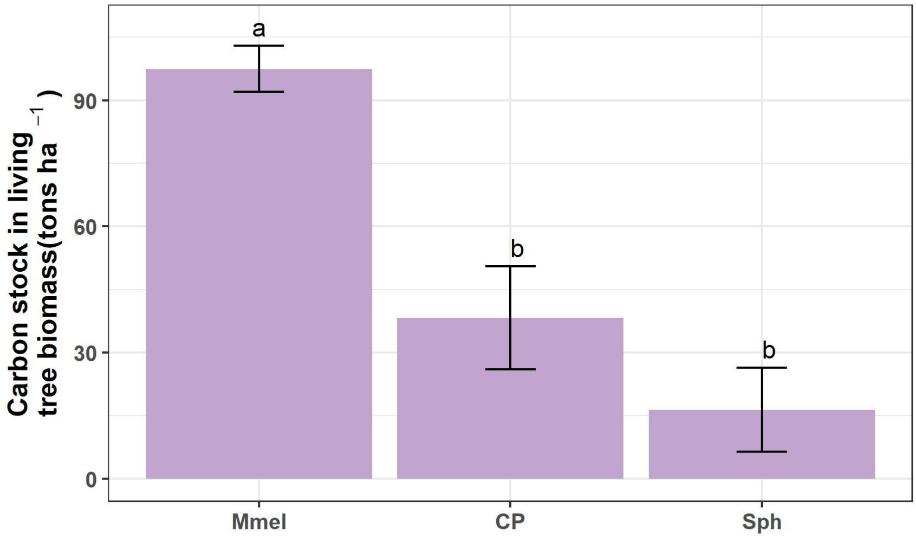

Figure 4. Carbon stock of living tree biomass. Mmel, *Myrtillosa* turf. mel. sites (n = 20); CP, *Caricoso–phragmitosa* sites (n = 4); Sph, *Sphagnosa* sites (n = 6). Different letters above bars represent significant differences ($p < 0.05$) in estimated means, as determined using Tukey's HSD test.

At the Mmel sites, the carbon stock in total living tree biomass was 97.48 ± 27.41 tons ha^{-1}. At the CP sites, it was 38.27 ± 14.67 tons ha^{-1}, and at the Sph sites, it was 16.39 ± 16.12 tons ha^{-1}. The highest carbon stock was found in the stems, and the lowest was found in the dead branches (Table 2). At the Mmel sites, the mean (mean ± SD) tree stem carbon stock was 53.54 ± 15.14 tons ha^{-1}; at the CP sites, it was 20.94 ± 10.95 tons ha^{-1}, and at the Sph sites, it was 6.08 ± 6.94 tons ha^{-1}.

Carbon accumulation in the soil was calculated using carbon content and soil density. Significant differences in estimated means were detected between the soil depths and sites (Figure 5). At soil depths of 0–10 and 10–20 cm, the carbon stock was similar between the forest sites (Table 2); however, significantly higher ($p < 0.01$) soil carbon stock was observed at the drained sites compared to the wet sites (CP, Sph). At greater depths (20–80 cm), the average carbon stock per 10 cm soil layer was lower compared to at 0–20 cm depth, but the differences were significant only between the Mmel and Sph sites. The linear mixed effect model for soil carbon stock determined that forest site type, soil depth, and their interaction were significant factors (Tables S3 and S4). The proportion of variance explaining the fixed effects was high (marginal R^2: 0.89).

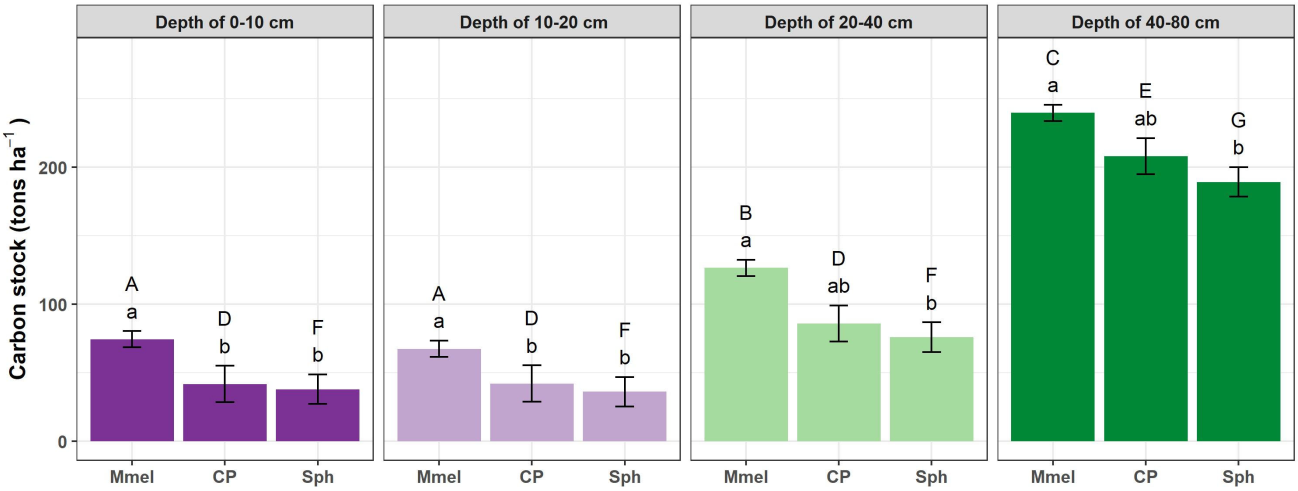

Figure 5. Soil carbon stock at different soil depths. Mmel, *Myrtillosa* turf. mel. sites; CP, *Caricoso–phragmitosa* sites; Sph, *Sphagnosa* sites. Different letters above bars represent significant differences ($p < 0.05$) in estimated mean carbon stock, as determined using Tukey's HSD test. Lowercase letters indicate significant differences between forest sites within each depth; uppercase letters indicate significant differences between soil depths within forest sites.

4. Discussion

Higher values of the forest inventory parameters (mean DBH, mean tree height, mean basal area, and mean volume) were found at the drained sites (Mmel) compared with the CP and Sph sites, resulting in higher tree biomass and tree CS (Table 2, Figure 4), thus validating our second hypothesis. This can be explained by the water table level (Figure 1). The water table level was higher at the Sph and CP sites, which limits oxygen availability for roots and impairs tree growth [7]. Enhanced tree growth and yield as a result of drainage has been reported in previous studies [42,43], but the results for these parameters vary between study sites [44]. In order to make generalized predictions and numerical estimates of additional growth induced by drainage in peatland forests, further studies with an expanded network of sample plots are needed due to various factors, such as ditch network design, site and environmental conditions, and stand history.

Soil penetration resistance was higher overall for the Sph and CP sites compared to the Mmel sites; however, the difference in penetration resistance was more distinct in the 0–20 cm layer than in deeper soil layers (Figure 2). Moreover, soil subsidence is a commonly reported finding in drained organic soils [18,45,46]. A study in Finland reported similar long-term soil subsidence values of 22 to 25 cm [45,46], which is in accordance with our observed soil subsidence results (Figure 3).

In the boreal biome, forest soils hold more carbon than overstories. Our results show that the soil carbon stock among the analyzed groups differed only between the Mmel and Sph sites ($p < 0.05$) when evaluated by the entire soil layer at a depth of 0–80 cm, thus validating our first hypothesis, as drained organic soil stores more carbon than undrained soil. Increased carbon stock in soil and trees has been reported in various studies of hemiboreal and boreal regions [26,46]. Previous studies have also shown that drainage systems can have both positive and negative impacts on greenhouse gas (GHG) emissions and removal at the landscape level [15–17], but the results may not be comparable due to the diverse stand factors. Climate conditions and variables are the main factors determining whether drained organic soil will be a source or a sink, and in boreal forests, drained organic soils mainly continue to act as sinks [19,46]. Therefore, it is important to consider these factors and conduct studies that are specific to the conditions of each individual site.

Our results show that the soil depth had a stronger effect on soil carbon stock than the analyzed site (Figure 5). Carbon stock increased with soil depth, which can be explained by the vertical movement of easily dissolvable organic compounds. After drainage, the litter decomposition of organic matter on the soil surface accelerates, and C from litter decomposition penetrates into and is subsequently stored in the deeper layers of the soil [47,48]. Litter production (needles, small twigs) plays an important role in the enrichment of carbon storage, which may offset the increased decomposition of soil organic matter after drainage [26,49,50]. In our case, the control sites did not have litter production because of weak tree growth. At the sample sites, litter provided a small amount of carbon stock. However, changes in hydrology due to drainage (lower water table) have an impact on the depth of aerated peat as well as the activity of microorganisms and oxidative processes in the soil, and improved water quality in the form of water flow can increase the availability of cleaner water [12,19]. After drainage, the increased decomposition of organic matter increases plant biomass and primary production; thus, a somewhat higher inflow of C into the ecosystem occurs [23,45].

The long-term impacts of forest drainage depend on various factors, including the intensity of drainage, the properties of the peatland, and the management practices employed. We have shown that over a long period (54 years after drainage), stand biomass (aboveground and belowground) at the Mmel sites compensated for soil CO_2 emissions. Our results suggest that the drained sites did not become a source of CO_2 emissions because CO_2 emissions do not overrun CO_2 assimilation. Similar results were reported in studies in boreal zones in Finland, where the authors concluded that the assimilation of poor drained peat soil carbon exceeded the emissions [12,18,19]. Furthermore, some studies have shown that rich drained peat soils do not become a source of GHGs after drainage [51]. We did not include carbon stock from understory vegetation such as shrubs, herbaceous plants, and mosses in our analysis. However, most of the changes in understory vegetation occur after drainage, as the change from plants adapted to waterlogged conditions to plants that favor drier conditions mostly affects root biomass, according to a study by Murphy, and does not significantly affect the total carbon stock and changes after drainage [52].

In the eastern Baltic region, the drainage of peat-forming ecosystems via ditching is a common practice [2,3], but there is a lack of data about the long-term monitoring of amelioration systems. In order to evaluate CS changes in drained organic soils, ground surface height measurements and soil carbon accumulation analysis should be repeated in a sufficiently large number of forest stands to characterize the different growth conditions and the possible initial state of forest stands. Methodologically, this task could be addressed by selecting new research objects in forest stands in peat forest massifs with drainage systems built 40–50 years ago. Such findings, in combination with the current knowledge, would be valuable for forest management and policy recommendations and help to optimize the management practices for climate change mitigation with the aim of maximizing carbon sequestration as well as reaching and sustaining climate neutrality.

5. Conclusions

The long-term drainage of organic soils in forests resulted in significantly higher tree carbon stock, and drainage did not deplete the soil carbon stock over a 54-year period. However, the effect of drainage was soil subsidence by an average of 25 cm and lower soil penetration resistance in the upper soil layer (0–20 cm). The drainage of organic soils in wet forests can favor climate change mitigation via increased carbon sequestration in tree biomass without depleting soil carbon stock; however, such areas have to be established in conjunction with other ecosystem services to ensure sustainable and multifunctional management at the landscape level. Moreover, further studies are needed to evaluate forests with very poor organic soils to supplement the current knowledge and evaluate the impact of fertility on carbon stock and budget in relation to management practices. The results of this study would be useful for management plans and policy recommendations for spatial planning for the allocation of stands with various targets to ensure that society's needs are met, climate change mitigation efforts are effective, and carbon sequestration is maximized.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/su152416622/s1, Table S1: Results of linear mixed effect model of soil penetration resistance; Table S2: Factors affecting soil penetration resistance based on analysis of variance (ANOVA) results; Table S3: Results of linear mixed effect model of soil carbon stock; Table S4: Factors affecting soil carbon stock based on analysis of variance (ANOVA) results.

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