PROMOCIJAS DARBA KOPSAVILKUMS
zinātnes doktora grāda
zinātnes doktors (Ph.D.) lauksaimniecības,
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SILTUMNĪCEFEKTA GĀZU EMISIJU IETEKMĒJOŠIE FAKTORI EITROFOS PURVAIŅOS UN KŪDREŅOS

Aldis Butlers

GREENHOUSE GAS EMISSIONS AND AFFECTING FACTORS IN FORESTS WITH NATURALLY WET AND DRAINED NUTRIENT-RICH ORGANIC SOILS

SUMMARY OF THE DOCTORAL THESIS
for the doctoral degree
Doctor of Science (Ph.D.)
in Agriculture, Forestry and Fisheries

LATVIJAS VALSTS MEŽZINĀTNES INSTITŪTS “SILAVA”
LATVIAN STATE FOREST RESEARCH INSTITUTE “SILAVA”

LATVIJAS BIOZINĀTŅU UN TEHNOLOĢIJU UNIVERSITĀTE
LATVIA UNIVERSITY OF LIFE SCIENCES AND TECHNOLOGIES

Mg.silv. **ALDIS BUTLERS**

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Dr.silv. Andis LAZDIŅŠ

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ANOTĀCIJA

Pētījuma aktualitāti nosaka Parīzes nolīgums un saistīti starptautiski normatīvie akti, paredzot, ka pēc 2050. gada zemes izmantošanas, zemes izmantošanas maiņas un mežsaimniecības (ZIZIMM) sektoram jākompensē Latvijas kopējās siltumnīcefekta gāzu (SEG) emisijas. Organiskā meža augsne (Latvijā tipiski kūdras un kūdrainās augsnes) ir būtisks SEG emisiju avots Latvijas mērogā, un vieni no efektīvākajiem klimata pārmaiņu mazināšanas pasākumiem ZIZIMM sektorā saistīti ar tās apsaimniekošanu. Tomēr joprojām trūkst zināšanu, lai novērtētu mežu ar dažādas auglības organisko augsni apsaimniekošanas scenāriju potenciālo ieguldījumu klimata pārmaiņu mazināšanā. Nacionālā SEG inventarizācijā hidromeliorētas organiskas augsnes (neatkarīgi no tās auglības) oglekļa dioksīda (CO_2) emisiju aprēķināšanai tiek piemērots viens nacionālos pētījumos iegūts CO_2 emisiju aprēķina faktors (EF). Savukārt, metāna (CH_4) un dislāpekļa oksīda (N_2O) emisiju aprēķināšanai tiek pielietots nacionālos apstākļos neverificēti EF, kas izstrādāti pētījumos mērenā klimata joslā. Pētījums īstenots, lai izstrādātu eitrofu kūdreņu un purvaiņu augsnes SEG EF un novērtētu meža ekosistēmas neto SEG emisijas. Iegūtās zināšanas pielietojamas nacionālās SEG inventarizācijas metodikas pilnveidošanā un klimata pārmaiņu mazinošo pasākumu plānošanā.

Empīriskais materiāls, augsnes SEG emisiju un oglekļa (C) ieneses raksturošanai, 12 mēnešu ilgā monitoringa laikā ievākts 31 meža nogabalā ar izcirtumiem un mežaudzēm (dumbrājs, liekņa, platlapju kūdrenis) dažādās attīstības stadijās. Augsnes CO_2, CH_4 un N_2O emisiju un C ieneses ar koku vainagu nobirām mērījumi piecos atkārtojumos katrā nogabalā veikti ar intervālu četras nedēļas. Vienlaicīgi ar SEG mērījumiem noteikta arī augsnes un gaisa temperatūra, kā arī gruntsūdens līmenis. C ienese ar zemsedzes veģetāciju un kokus sīksaknēm novērtēta, veicot to biomasas mērījumus veģetācijas sezonas beigās. Augsnes C uzkrājuma izmaiņas aprēķinātas summējot novērtētās gada kumulatīvās augsnes CO_2-C emisijas un C ienesi. Novērtētās sakarības starp augsnes SEG emisijām, C ienesi un ietekmējošiem faktoriem pielietotas, lai kvantificētu ekosistēmas ikgadējo neto SEG emisiju dinamiku apsaimniekotos mežos, novērtējumā ietverot arī ikgadējo C piesaisti dzīvā un nedzīvā koksnes biomasā, nocirstas koksnes produktos un biokurināmā aizvietošanas efektu.

Novērtētās ikgadējās augsnes bruto CO_2 emisijas izcirtumos ($7{,}70 \pm 0{,}53$ t C ha^{-1} gadā) ir būtiski lielākās nekā mežaudzēs ($6{,}14 \pm 0{,}15$ t C ha^{-1} gadā). Meža apsaimniekošanas cikla laikā eitrofu kūdreņu un purvaiņu augsnes ikgadējā neto CO_2 piesaiste ir attiecīgi vidēji $0{,}28 \pm 0{,}66$ t C ha^{-1} gadā un $0{,}42 \pm 0{,}43$ t C ha^{-1} gadā. Mežaudzēs galvenie augsnes C ieneses avoti ir zemsedzes veģetācija un koku vainagu nobiras, nodrošinot attiecīgi vidēji $41 \pm 8\%$ un $43 \pm 6\%$ no pētījumā novērtētās kopējās augsnes C ieneses. Apsaimniekoti eitrofi purvaiņi piesaista vidēji $0{,}2 \pm 9{,}7$ t CO_2 ekv. gadā, bet eitrofi kūdreņi – vidēji $2{,}9 \pm 14{,}4$ t CO_2 ekv. gadā.

ABSTRACT

The topicality of this study is determined by the Paris Agreement and related international regulatory acts, which stipulate that after 2050, the land use, land use change, and forestry (LULUCF) sector must compensate for Latvia's total greenhouse gas (GHG) emissions. Organic forest soils, particularly peat and peaty soils in Latvia, are a significant source of GHG emissions in the country, and one of the most effective climate change mitigation measures in the LULUCF sector is related to their management. However, there is currently a lack of knowledge on the potential contribution of forests with different nutrient availability organic soil management scenarios to mitigating climate change. In the national GHG inventory, a single carbon dioxide (CO_2) emission factor (EF) obtained from national studies is applied to calculate the CO_2 emissions from drained organic soil, regardless of its nutrient availability. For the calculation of methane (CH_4) and nitrous oxide (N_2O) emissions, unverified EFs developed in studies in a temperate climate zone are used in the national inventory. This study aims to develop GHG EFs for drained and undrained nutrient-rich organic forest soils and to estimate the net GHG emissions of the forest ecosystem with such soils. The acquired knowledge can be used to improve the national GHG inventory methodology and to plan climate change mitigation measures.

Empirical material for characterizing soil GHG emissions and soil C input was collected during a 12-month monitoring period in 31 forest compartments with clearcuts and forest stands in various stages of development. Measurements of soil CO_2, CH_4, and N_2O emissions, as well as soil C input by foliar litter, were carried out in five replicates in each plot with an interval of four weeks. Simultaneously with the GHG measurements, soil and air temperature, as well as groundwater level, were also determined. Soil C input by ground vegetation and fine roots of trees was estimated by biomass measurements at the end of the growing season. Changes in soil C stock were calculated by summing the estimated annual cumulative soil CO_2-C emissions and C input. The evaluated relationships between soil GHG emissions, C input, and affecting factors were used to quantify the dynamics of the ecosystem's annual net GHG emissions in managed forests, by taking into account also the annual C sequestration in living biomass and deadwood, harvested wood products, and the biofuel replacement effect.

The estimated annual gross soil CO_2 emissions in clearcuts (7.70 ± 0.53 t C ha^{-1} year^{-1}) are significantly higher than in forest stands (6.14 ± 0.15 t C ha^{-1} year^{-1}). During the forest management cycle, the annual net CO_2 sequestration by nutrient-rich drained and undrained forest soils is on average 0.28 ± 0.66 t C ha^{-1} year^{-1} and 0.42 ± 0.43 t C ha^{-1} year^{-1}, respectively. In forest stands, the main sources of soil C input are ground vegetation and foliar litter, providing an average of 41 ± 8% and 43 ± 6% of the total soil C input estimated in the study, respectively. Managed forests with undrained and drained nutrient-rich soil sequester an average of 0.2 ± 9.7 and 2.9 ± 14.4 t CO_2 eq. year^{-1}, respectively.

SATURS / *CONTENTS*

PROMOCIJAS DARBĀ LIETOTIE SAĪSINĀJUMI

B – bērzs (*Betula pendula*)
C – ogleklis
CH_4 – metāns
CO_2 – oglekļa dioksīds
E – egle (*Picea abies*)
EC – mikrometeoroloģiskie mērījumi (*Eddy-Covariance*)
EF – emisiju aprēķina faktors
GŪ – gruntsūdens
IPCC – Apvienoto Nāciju Organizācijas Klimata pārmaiņu starpvaldību padome
LVGMC – Latvijas Vides ģeoloģijas un meteoroloģijas centrs
Ma – melnalksnis (*Alnus glutinosa*)
MRM – Meža resursu monitorings jeb Meža statistiskā inventarizācija
MAAT – meža augšanas apstākļu tips
MT – meža tips
N – slāpeklis
N_2O – dislāpekļa oksīds
NEE – neto ekosistēmas C apmaiņa
NEP – neto ekosistēmas produktivitāte
Organiskā augsne – kūdras un kūdrainās augsnes atbilstoši IPCC definīcijai
PCA – Principālo komponenšu analīze
pZV – zemsedzes veģetācijas saknes
Rhet – augsnes heterotrofā elpošana
Rkop – augsnes kopējā elpošana (augsnes heterotrofās un veģetācijas, tajā skaitā gan virszemes, gan sakņu, autotrofās elpošanas summa)
SEG – siltumnīcefekta gāzes
SEG inventarizācija – nacionālo antropogēno siltumnīcefekta gāzu emisiju novērtējums atbilstoši Apvienoto Nāciju Organizācijas Vispārējās konvencijas par klimata pārmaiņām, Eiropas Parlamenta un Padomes normatīvo aktu un Klimata pārmaiņu starpvaldību padomes metodisko norādījumu prasībām
Ta – augsnes temperatūra 5 cm dziļumā
VKS – valdošā koku suga
vZV – virszemes zemsedzes veģetācija
ZIZIMM – SEG inventarizācijas zemes izmantošanas, zemes izmantošanas maiņas un mežsaimniecības sektors
ZV – zemsedzes veģetācija

1. DARBA VISPĀRĪGS RAKSTUROJUMS

1.1. Temata aktualitāte

Pētījuma aktualitāti nosaka Parīzes nolīgums un saistīti starptautiski normatīvie akti, paredzot, ka pēc 2050. gada zemes izmantošanas, zemes izmantošanas maiņas un mežsaimniecības (ZIZIMM) sektoram jākompensē Latvijas kopējās siltumnīcefekta gāzu (SEG) emisijas. Organiskā meža augsne (atbilstoši Klimata pārmaiņu starpvaldību padomes definīcijai Latvijā tipiski kūdras un kūdrainās augsnes) ir būtisks SEG emisiju avots Latvijas mērogā, un vieni no efektīvākajiem klimata pārmaiņu mazināšanas pasākumiem ZIZIMM sektorā saistīti ar tās apsaimniekošanu. Tomēr joprojām trūkst zināšanu, lai novērtētu mežu ar dažādas auglības organisko augsni apsaimniekošanas scenāriju potenciālo ieguldījumu klimata pārmaiņu mazināšanā. Nacionālā SEG inventarizācijā hidromeliorētas organiskas augsnes (neatkarīgi no tās auglības) oglekļa dioksīda (CO_2) emisiju aprēķināšanai tiek piemērots viens nacionālos pētījumos iegūts CO_2 emisiju aprēķina faktors (EF). Savukārt, metāna (CH_4) un dislāpekļa oksīda (N_2O) emisiju aprēķināšanai tiek pielietots nacionālos apstākļos neverificēti EF, kas izstrādāti pētījumos mērenā klimata joslā. Pētījums īstenots, lai izstrādātu eitrofu kūdreņu un purvaiņu augsnes SEG EF un novērtētu meža ekosistēmas neto SEG emisijas. Iegūtās zināšanas pielietojamas nacionālās SEG inventarizācijas metodikas pilnveidošanā un klimata pārmaiņu mazinošo pasākumu plānošanā.

1.2. Promocijas darba mērķis, uzdevumi, tēzes

<u>Promocijas darba mērķis</u> ir novērtēt eitrofu kūdreņu un purvaiņu augsnes un ekosistēmas kopējās siltumnīcefekta gāzu (CO_2, CH_4, N_2O) emisijas.

Mērķa sasniegšanai izvirzīti sekojoši <u>pētnieciskie uzdevumi</u>:

1. izstrādāt koeficientus, kas raksturo oglekļa ienesi augsnē ar koku vainaga nobirām, koku sīksaknēm un zemsedzes veģetāciju eitrofos egļu (*Picea abies* (L.) H. Karst), bērzu (*Betula* spp.) un melnalkšņu (*Alnus glutinosa* (L.) Gaertn) kūdreņos un purvaiņos;
2. izstrādāt augsnes CO_2, N_2O un CH_4 emisiju aprēķina faktorus eitrofiem egļu, bērzu un melnalkšņu kūdreņiem un purvaiņiem;
3. novērtēt eitrofu egles, bērzu un melnalkšņa kūdreņu un purvaiņu kopējās siltumnīcefekta gāzu emisijas.

<u>Pētījumā izvirzītas sekojošas tēzes</u>:

1. Eitrofos purvaiņos un kūdreņos augsne nezaudē tās oglekļa uzkrājumu.
2. Eitrofu purvaiņu un kūdreņu ekosistēmas nav siltumnīcefekta gāzu emisiju avots.

1.3. Darba zinātniskā novitāte un praktiskā nozīme, rekomendācijas

Līdzšinējie augsnes SEG emisiju pētījumi apsaimniekotos mežos galvenokārt īstenoti hidromeliorētās platībās, un rezultāti nenodrošina vienmērīgu ģeogrāfisko reprezentativitāti. Lielākā daļa no pētījumu objektiem atrodas Somijā, un tie ir ierīkoti hidromeliorētos mežos ar atšķirīgas auglības organisko augsni. Savukārt, pētījumi par SEG emisijām hemiboreālos mežos ir nepietiekami. Turklāt, pētījumi galvenokārt īstenoti, novērtējot augsnes SEG emisijas vai neto oglekļa (C) uzkrājuma izmaiņas tikai pētījuma monitoringa periodā. Tādējādi iztrūkst zināšanas par augsnes C uzkrājuma un ekosistēmas SEG emisiju dinamiku meža apsaimniekošanas cikla laikā. Iztrūkst arī zināšanas par organiskās augsnes ar saglabātu hidroloģisko režīmu (nav veikta hidromeliorācija) SEG emisijām. Jo nosacījums, ka SEG inventarizācijā nav atsevišķi jānovērtē organiskās augsnes ar saglabātu hidroloģisko režīmu SEG emisijas apsaimniekotos mežos, nav motivējis šādas augsnes SEG emisiju novērtēšanu līdzšinējos pētījumos. SEG inventarizācijā pielietotā pieeja – novērtēt SEG emisijas tikai no hidromeliorētas organiskām augsnēm – nesniedz pilnīgu izpratni par meža organiskās augsnes hidromeliorēšanas vai dabiskā hidroloģiskā režīma saglabāšanas kvantitatīvu ietekmi uz valsts kopējām SEG emisijām. Tādēļ apzināt emisijas, ko rada organiskā augsne ar saglabātu hidroloģisko režīmu ir tikpat būtiski kā novērtēt hidromeliorētas augsnes radītās emisijas. Promocijas darbā veiktā pētījuma rezultāti risina minēto zināšanu trūkuma problēmu.

Pētījumā kvantificēta purvaiņu un kūdreņu augsnes, kā arī ekosistēmas kopējo neto SEG emisiju dinamika meža apsaimniekošanas cikla laikā. Iegūtās zināšanas ļauj salīdzināt meža apsaimniekošanas scenāriju, ar un bez meža hidromeliorācijas, ietekmi uz ekosistēmas SEG emisijām. Pētījumā izstrādātie SEG emisiju aprēķina faktori un vienādojumi pielietojami hemiboreālo mežu SEG inventarizācijas metodikas pilnveidošanai.

Rekomendācijas:

1. Auglīgas organiskās augsnes ikgadējās C ieneses aprēķināšanai ieteicams izmantot sekojošos pētījuma rezultātus:
   a. regresijas vienādojumus, kas raksturo lapkoku vai egļu meža vainaga nobiru C ienesi atkarībā no mežaudzes šķērslaukuma;
   b. zemsedzes veģetācijas biomasas C ieneses koeficientus kūdreņos un purvaiņos ar valdošo koku sugu bērzs, melnalksnis un egle;
   c. ikgadējā koku sīksakņu biomasas atmiruma koeficientus lapkoku un egļu mežos atkarībā no augsnes hidroloģiskā režīma.

   Augsnes C ieneses prognozēšanas spēju pilnveidošanai un prognožu rezultātu nenoteiktības novērtēšanai Latvijas mēroga aprēķiniem nepieciešami ekstensīvi ilgtermiņa pētījumi par augsnes oglekļa ienesis ar vainaga nobirām (to frakciju sadalījumā), koku sīksaknēm un zemsedzi un tās variāciju atkarībā no ikgadēji mainīgiem meteoroloģiskiem apstākļiem.
2. Augsnes N_2O emisiju aprēķināšanai pielietojami pētījumā izstrādātie emisiju aprēķina faktori. CH_4 emisijas ieteicams aprēķināt, izmantojot regresijas

vienādojumu, kas raksturo emisijas atkarībā no vidējā gruntsūdens līmeņa platībā un ņem vērā arī ekstrēmo emisiju sastopamības varbūtību. Lai izvairītos no CH_4 emisiju pārvērtēšanas ziemas periodā, augsnes temperatūra –5°C pielietojama kā robežvērtība, kuru pārsniedzot, emisijas pieņemamas kā nebūtiskas. CH_4 emisiju aprēķina rezultāta nenoteiktība strauji pieaug, ja vidējais gruntsūdens līmenis platībā ir seklāk par 30 cm, tādēļ nenoteiktības mazināšanai plašāki pētījumi turpināmi platībās ar vidējo gruntsūdens līmeņa dziļumu 0 līdz 30 cm.

3. Ikgadējās gaisa temperatūras mainība gada kumulatīvās augsnes kopējās elpošanas CO_2 emisijas var ietekmēt par vidēji ± 1,6 t CO_2-C ha^{-1} gadā, tādēļ, prognozējot augsnes emisijas valsts mērogā un ilgtermiņā, jāņem vērā reģionālo un ikgadējo gaisa temperatūru mainība. Gada kumulatīvo augsnes elpošanas emisiju aprēķinā svarīgi izvēlēties korektu interpolācijas pieeju. Ieteicams emisijas interpolēt atkarībā no stundas vidējās gaisa temperatūras, jo gada kopējo augsnes elpošanu aprēķinot pēc diennakts vidējās gaisa temperatūras, tās var potenciāli tikt pārvērtētā par vidēji 1,50 t CO_2-C ha^{-1} gadā. Analoģiski, prognozējot CH_4 emisijas, izvērtējama ikgadējo meteoroloģisko apstākļu ietekme uz ikgadējo vidējo gruntsūdens līmeni.
4. Ikgadējo meža ekosistēmas kopumā vai atsevišķi meža augsnes SEG emisijas un CO_2 piesaisti nosaka mežaudzes attīstības stadija, saimnieciskā darbība un meteoroloģiskie apstākļi. Šie un saistītie SEG emisiju ietekmējošie faktori jāņem vērā, novērtējot un salīdzinot mežu SEG emisijas dažādos to apsaimniekošanas scenārijos.

1.4. Zinātniskā darba publicitāte

<u>Zinātniskie raksti:</u>

I Vanags-Duka, M., Bārdule, A., **Butlers, A.**, Upenieks, E. M., Lazdiņš, A., Purviņa, D., & Līcīte, I. (2022). GHG Emissions from Drainage Ditches in Peat Extraction Sites and Peatland Forests in Hemiboreal Latvia. *Land*, 11(12), 2233; 10.3390/land11122233.

II **Butlers, A.**, Lazdiņš, A., Kalēja, S., & Bārdule, A. (2022). Carbon Budget of Undrained and Drained Nutrient-Rich Organic Forest Soil. *Forests*, 13, 1790; 10.3390/f13111790.

III Bārdule, A., Gerra-Inohosa, L., Kļaviņš, I., Kļaviņa, Z., Bitenieks, K., **Butlers, A.**, Lazdiņš, A., & Lībiete, Z. (2022). Variation in the Mercury Concentrations and Greenhouse Gas Emissions of Pristine and Managed Hemiboreal Peatlands. *Land*, 11(9), 1414; 10.3390/land11091414.

IV Lazdins, A., **Butlers, A.**, & Ancans, R. (2022). Nitrous Oxide (N_2O) and Methane (CH_4) Fluxes from Tree Stems in Birch and Black Alder Stands – a Case Study in Forests with Deep Peat Soils. In: Proceedings of 21st International Scientific Conference "Engineering for Rural Development", Jelgava, Latvia, 25–27 May 2022. Jelgava: LULST, p. 754–759; 10.22616/ERDev.2022.21.TF229.

V **Butlers, A.**, Spalva, G., Licite, I., & Purvina, D. (2022). Carbon Dioxide (CO_2) Emissions from Naturally Wet and Drained Nutrient-Rich Organic Forests Soils. In: Proceedings of 21st International Scientific Conference “Engineering for Rural Development”, Jelgava, Latvia, 25–27 May 2022. Jelgava: LULST, p. 577–582; 10.22616/ERDev.2022.21.TF19.

VI **Butlers, A.**, Bārdule, A., Spalva, G., & Muižnieks, E. (2021). N_2O and CH_4 Emissions from Naturally Wet and Drained Nutrient-Rich Organic Forest Soils. In: Proceedings of the 10th International Scientific Conference “Rural Development 2021: Challenges for Sustainable Bioeconomy and Climate Change”, Kaunas, Lithuania, 21–23 September 2021. Kaunas: Vytautas Magnus University Agriculture Academy, p. 195–200; 10.15544/RD.2021.030.

VII Bārdule, A., **Butlers, A.**, Lazdiņš, A., Līcīte, I., Zvirbulis, U., Putniņš, R., Jansons, A., Adamovičs, A., & Razma, Ģ. (2021). Evaluation of Soil Organic Layers Thickness and Soil Organic Carbon Stock in Hemiboreal Forests in Latvia. *Forests*, 12(7), 840; 10.3390/f12070840.

VIII Bārdule, A., Liepiņš, J., Liepiņš, K., Stola, J., **Butlers, A.**, & Lazdiņš, A. (2021). Variation in Carbon Content among the Major Tree Species in Hemiboreal Forests in Latvia. *Forests*, 12(9), 1292; 10.3390/f12091292.

IX Bārdule, A., Petaja, G., **Butlers, A.**, Purviņa, D., & Lazdiņš, A. (2021). Estimation of Litter Input in Hemi-Boreal Forests with Drained Organic Soils for Improvement of GHG Inventories. *Baltic Forestry*, 27(2), 534; 10.46490/BF534.

<u>Dalība konferencēs</u>:

1. 21th International Scientific Conference “Engineering for Rural Development”, 25.–27.05.2022., Jelgava, Latvija. Prezentācija – **Butlers, A.**, Spalva, G., Līcīte, I., & Purviņa, D. Carbon dioxide (CO_2) emissions from naturally wet and drained nutrient-rich organic forests soils.
2. 21th International Scientific Conference “Engineering for Rural Development”, 25.–27.05.2022., Jelgava, Latvija. Prezentācija – Lazdiņš, A., **Butlers, A.**, & Ancāns, R. Nitrous oxide (N_2O) and methane(CH_4) fluxes from tree stems in birch and black alder stands – a case study in forests with deep peat soils.
3. Annual 26th International Scientific Conference “Research for Rural Development”, 18.–20.05.2022., Jelgava, Latvija. Prezentācija – **Butlers, A.**, & Lazdiņš, A. Latvia case study of greenhouse gas (GHG) fluxes from flooded former peat extraction fields in central part of Latvia.
4. 10th International Scientific Conference “Rural Development 2021: Challenges for Sustainable Bioeconomy and Climate Change”, 21.–23.09.2021., Kauņa, Lietuva. Prezentācija – **Butlers, A.**, Bārdule, A., Spalva, G., & Muižnieks, E. N_2O and CH_4 emissions from naturally wet and drained nutrient-rich organic forest soils.
5. 20th International Scientific Conference “Engineering for Rural Development”, 26.–28.05.2021., Jelgava, Latvija. Prezentācija – Lazdiņš, A., Šņepsts, G.,

Butlers, A., Purviņa, D., Zvaigzne, A. Z., & Līcīte, I. Evaluation of middle term greenhouse gas (GHG) mitigation potential of birch plantations with mineral and organic soils.

6. Konference “Zināšanās balstīta meža nozare”, 27.01.2021., tiešsaiste. Prezentācija – **Butlers, A.** Siltumnīcefekta gāzu emisijas ietekmējošie faktori mežos ar dabiski mitrām un meliorētām auglīgām organiskām augsnēm.
7. Annual 26th International Scientific Conference “Research for Rural Development”, 12.05.2020., Jelgava, Latvija. Prezentācija – **Butlers, A.**, & Lazdiņš, A. Carbon stock in litter and organic soil in drained and naturally wet forest lands in Latvia.

1.5. Promocijas darba struktūra un apjoms

Promocijas darba struktūra ir pakārtota darbā izvirzītajiem pētnieciskajiem uzdevumiem. Darbs sastāv no trīs nodaļām. Pirmajā nodaļā veikts līdzšinējo zināšanu par meža organiskās augsnes SEG emisijām un C aprites izvērtējums. Otrajā nodaļā aprakstītas darbā izmantotās empīriskā materiāla iegūšanas un apstrādes metodes. Trešajā nodaļā aprakstīti un izvērtēti pētījumā iegūtie rezultāti, atbilstoši izvirzītajam promocijas darba mērķim un pētnieciskajiem uzdevumiem.

Promocijas darba apjoms ir 106 lpp., 19 tabulas, 39 attēli, 5 pielikumi un 296 literatūras avoti. Atbilstoši galvenajiem pētījuma rezultātiem formulēti deviņi secinājumi un sniegtas četras rekomendācijas.

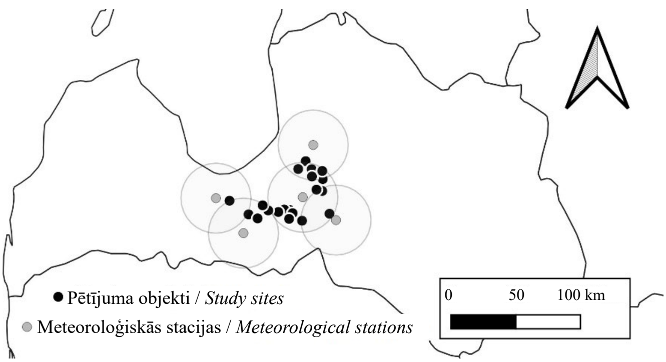
2. MATERIĀLS UN METODIKA

Empīriskais materiāls ievākts 31 meža nogabalā laika posmā no 2019. gada oktobra līdz 2022. gada maijam. Katrs meža nogabals pārstāvēts ar vienu parauglaukumu (2.1. att.), kuros veikti kokaudžu raksturlielumu, gruntsūdens (GŪ) līmeņa dziļuma, augsnes un atmosfēras temperatūras mērījumi, kā arī ievākti augsnes SEG emisiju, augsnes, augsnes ūdens, vainaga nobiru, zemsedzes veģetācijas un koku sīksakņu biomasas paraugi to analizēšanai laboratorijā. Katrā parauglaukumā empīriskais materiāls ievākts 12 secīgu mēnešu periodā.

Vainaga nobiru, zemsedzes veģetācijas un koku sīksakņu biomasas paraugi ievākti un noteikta to sausne un C saturs, lai raksturotu ikgadējo augsnes C ienesi un tā uzkrājuma līdzsvaru augsnē. Iegūto augsnes SEG emisiju un C ieneses augsnē, kā arī to ietekmējošo faktoru mērījumu rezultāti izmantoti SEG emisiju no mežiem ar auglīgu organisko augsni ietekmējošo faktoru un to savstarpējo sakarību identificēšanai un raksturošanai.

2.1. Pētījuma objektu raksturojums

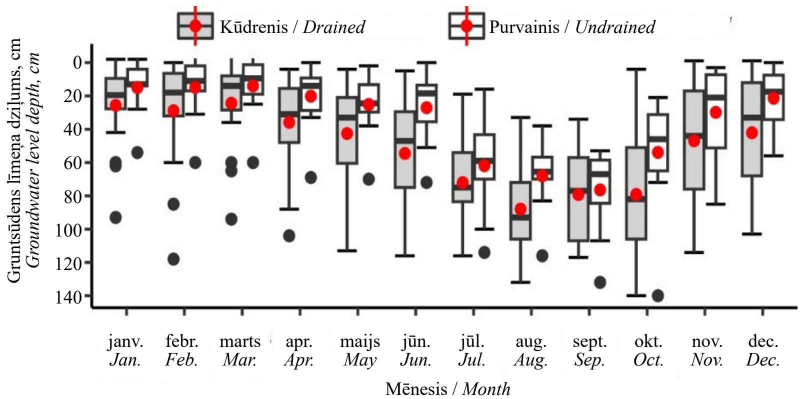
Lai raksturotu SEG emisijas un to ietekmējošos faktorus mežos ar auglīgu organisko augsni, ierīkoti parauglaukumi un empīriskais materiāls ievākts 21 platlapju kūdreņu (*Oxalidosa turf. mel.*) un 10 dumbrāju (*Dryopterioso-caricosa*) vai liekņu (*Filipendulosa*) mežos dažādās attīstības stadijās. Pētījumā meži ar saglabāta hidroloģiskā režīma augsni (dumbrāji un liekņas) un hidromeliorētu augsni (platlapju kūdreņi) pārstāvēti ar 10 līdz 80 gadus vecām mežaudzēm (kopā 26) un 5 izcirtumiem. Katrs ierīkotais parauglaukums atrodas ne vairāk kā 30 km attālumā no kādas no vistuvāk esošajām Latvijas Vides, ģeoloģijas un meteoroloģijas centra (LVĢMC) meteoroloģiskajām stacijām (2.1. att.). Platlapju kūdreņi (Kp) pārstāvēti ar trīs bērzu (*Betula pendula*), diviem melnalkšņa (*Alnus glutinosa*), 12 egļu (*Picea abies*) audzēm un četriem izcirtumiem, savukārt, dumbrāji (Db) un liekņas (Lk) pārstāvēti



2.1. att. **Parauglaukumu atrašanās vietas un meteoroloģiskās stacijas**
Fig. 2.1. ***Locations of sample plots and closest meteorological stations***

ar trīs bērzu (B), 5 melnalkšņa (Ma), vienu egles (E) audzi un vienu izcirtumu. Pētījuma objektu raksturošanai parauglaukumos ievākti un laboratorijā analizēti augsnes un augsnes ūdens paraugi, kā arī veikti GŪ līmeņa mērījumi. Pētījuma mežaudzēs noteikti arī kokaudžu raksturlielumi. Pētījuma objektu raksturošanai iegūtie mērījumu un analīžu rezultāti izmantoti arī augsnes SEG emisiju ietekmējošo faktoru identificēšanai un sakarību raksturošanai.

GŪ līmenis no augsnes virsmas bijis vidēji 55 ± 2 cm dziļumā kūdreņos un 35 ± 3 cm – purvaiņos. Ikmēneša vidējais GŪ līmenis kūdreņos bijis par vidēji 18 ± 2 cm zemāk no augsnes virsmas, salīdzinot ar GŪ līmeni purvaiņos (2.2. att.).



2.2. att. **Gruntsūdens līmeņa dziļums pētījuma objektos**
Fig. 2.2. ***Groundwater level depth at the study sites***

Pētījuma objektos C saturs augsnes virsējā 20 cm slānī ir no 342 līdz 507 g C kg^{-1} (vidēji 455 ± 43 g C kg^{-1}) purvaiņos un 328 līdz 569 g C kg^{-1} (vidēji 487 ± 40 g C kg^{-1}) platlapju kūdreņos, savukārt, vidējā C/N attiecība neatkarīgi no MAAT ir 19 ± 3. Individuālos pētījuma objektos C/N attiecība ir no 13 līdz 31. Arī pārējo vērtēto augsnes ķīmisko elementu koncentrācijas, pH un augsnes blīvuma vidējās vērtības kūdreņos un purvaiņos būtiski neatšķiras. Vidējās ķīmisko parametru vērtības pētījuma objektos ir: 0,5 ± 0,1 g K kg^{-1}; 21 ± 4,5 g Ca kg^{-1}; 2,1 ± 0,4 g Mg kg^{-1} un 1,3 ± 0,4 g P kg^{-1}. Savukārt, vidējais augsnes blīvums ir 426,0 ± 29,3 kg m^{-3}, bet augsnes pH 4,5 ± 0,4.

2.2. Vainaga nobiru, zemsedzes veģetācijas un koku sīksakņu paraugu ievākšana un analīze

Koku vainagu nobiras (nobiras) uztvertas, izmantojot katrā parauglaukumā vienmērīgi izvietotus piecus konusa formas nobiru uztvērējus ar laukumu 0,5 m^2. Uztvertās nobiras ievāktas 12 secīgus mēnešus ar intervālu 4 nedēļas un nogādātas laboratorijā. Sausna noteikta katram ievāktajam paraugam. Visā 12 mēnešu periodā no viena nobiru uztvērēja ievāktās nobiras apvienotas un samaltas smalkā pulverī C satura noteikšanai. Sausnas noteikšanai svērts viss uztvertais ar koku nobirām saistītais materiāls, tajā skaitā skujas, lapas, koksne, miza, čiekuri, sēklas, ķērpji, izņemot zarus, garākus par 10 cm.

Zemsedzes veģetācijas paraugi ņemti katrā parauglaukumā četros atkārtojumos, atsevišķi ņemot zemsedzes veģetācijas (lakstaugi) virszemes un sakņu biomasas paraugus. Paraugošanai katrā parauglaukumā izvēlēts 1 m^2 kvadrāta formas laukums ar mežaudzei raksturīgu veģetāciju. Paraugi ievākti, kad sagaidāms zemsedzes biomasas daudzuma maksimums – augustā (Uri et al., 2017). Veģetācijas paraugi ņemti no 4 mazāka izmēra kvadrāta formas laukumiem (malas garums 20 cm) iepriekš izraudzītā 1 m^2 laukuma kvadrāta stūros. Virszemes biomasas paraugam līdz ar augsnes virsmu ievākta visa lakstaugu dzīvā veģetācija, bet sakņu biomasa paraugam – zemsedzes veģetācijas saknes no augsnes 20 cm virsējā slāņa. Laboratorijā sakņu biomasa noskalota (mitrā sijāšana), lai atbrīvotos no augsnes daļiņām un koku saknēm atbilstoši to morfoloģiskām pazīmēm.

Sīksakņu (diametrs < 2 mm) produkcijas paraugi ievākti, izmantojot modificētu sīksakņu ieaugšanas cilindra metodi (Laiho et al., 2014a; Bhuiyan et al., 2017). Metodes pamatā ir ar parauglaukumā iegūtu kūdru piepildīts elastīgs poliestera cilindra formas sietveida maiss (ieaugšanas cilindrs) ar garumu 80 cm, diametru 35 mm un acu izmēru 2 × 2 mm, kas ievietots kūdrā iespiestā, 60 cm dziļā caurumā. Katrā parauglaukumā pirms veģetācijas sezonas sākuma virzienā no parauglaukuma centra uz tā ārējo robežu ar intervālu viens metrs ierīkoti seši ieaugšanas cilindri. Augsne ieaugšanas cilindra piepildīšanai iegūta netālu no tā ierīkošanas vietas, izmantojot augsnes urbi. Puse ieaugšanas cilindru no augsnes izņemti pēc vienas veģetācijas sezonas noslēguma, bet atlikušie – pēc divām veģetācijas sezonām. Sīksakņu ieaugšanas cilindri no augsnes izņemti pēc iespējas izvairoties no ieaugušo sakņu izraušanas no cilindra, tās pirms tam apgriežot. Ieaugšanas cilindri nogādāti laboratorijā, kur ieaugušās saknes no ārpuses apgrieztas līdz ar cilindra virsmu, un cilindrā ietilpstošās sīksaknes atdalītas no augsnes, veicot slapjo sijāšanu. Pirms sīksakņu sausnas noteikšanas no izsijātā sakņu parauga pēc morfoloģiskām pazīmēm izšķirotas tikai koku saknes.

Nobiru, zemsedzes veģetācijas un koku sīksakņu paraugu biomasas sausnas saturs noteikts, paraugus žāvējot žāvskapī 70°C temperatūrā līdz nemainīgai masai un nosvērti. Pēc paraugu sausnas noteikšanas, smalkā pūderī samaltiem nobiru un zemsedzes veģetācijas paraugiem ar sausās sadedzināšanas metodi (elementanalīze) noteikts C saturs.

2.3. Augsnes SEG emisiju paraugu ņemšana un analīze

Augsnes SEG emisiju paraugu ņemšana īstenota, pielietojot manuālo slēgtas kameras metodi (Hutchinson & Livingston, 1993). Metodes īstenošanā pielietotā SEG gāzu emisiju paraugu ņemšanas komplekta galvenās komponentes ir augsnes gredzens un gāzu paraugu ņemšanas kamera (kamera) baltā krāsā no PVC materiāla. Augsnes gredzena diametrs – 50 cm – sakrīt ar kameras diametru, kuras augstums ir 40 cm un tilpums 63 L. Lai noteiktu SEG emisijas piecos atkārtojumos, katrā parauglaukumā līdz 5 cm augsnes dziļumā vismaz mēnesi pirms pirmās augsnes SEG emisiju paraugu ņemšanas ierīkoti pieci pastāvīgi augsnes gredzeni. Gredzeni ierīkoti, izvairoties no sakņu apgriešanas un saglabājot neskartu zemsedzes veģetāciju un zemsegas slāni (Pavelka et al., 2018), kas saglabāti neskarti arī visā augsnes SEG emisiju monitoringa īstenošanas laikā. Tādējādi ievāktie gāzu paraugi raksturo augsnes kopējo elpošanu (*Rkop*) – kamerā ietvertās augsnes heterotrofās elpošanas, kā arī veģetācijas virszemes un zemē esošās biomasas autotrofās elpošanas summu.

Augsnes SEG paraugi katrā parauglaukumā ņemti 12 secīgus mēnešus ar parauglaukumu apsekošanas un paraugu ņemšanas intervālu četras nedēļas. Paraugu ņemšanas laikā kamera tika novietota uz augsnes gredzena. Pirmais gāzu paraugs no kameras 100 mL stikla pudelēs ar 0,3 mbar retinājumu tika paņemts tūlītēji pēc kameras uzstādīšanas. Katrs nākamais gāzu paraugs no kameras ņemts ar 10 minūšu intervālu, līdz 30 minūšu laikā no kameras paņemti 4 paraugi. Pēc paraugu ņemšanas, paraugu pudeles nogādātas laboratorijā paņemto gāzu testēšanai ar gāzu hromatogrāfu. Gāzu saturs (CO_2, CH_4 un N_2O koncentrācija) ievāktajos augsnes SEG emisiju paraugos noteikts ar gāzu hromatogrāfu Shimadzu Nexis GC-2030 (Loftfield et al., 1997).

Vienlaicīgi ar augsnes SEG emisiju paraugu ņemšanas procedūras izpildi, veikti temperatūru un GŪ līmeņa dziļuma mērījumi. Temperatūra noteikta gaisam un augsnei 5 cm dziļumā netālu no augsnes gredzena (Pavelka et al., 2018). GŪ līmeņa dziļums noteikts ar mērlenti katrā parauglaukumā iepriekš ierīkotās GŪ līmeņa mērījuma akās (PVC caurule, kas ierīkota līdz 140 cm dziļumam).

2.4. Oglekļa ieneses augsnē ar zemsedzes veģetāciju, nobirām un sīksaknēm aprēķināšana

Ikgadējā C ienese augsnē aprēķināta ikgadējo nobiru, sīksakņu produkcijas vai zemsedzes veģetācijas neto ekosistēmas produktivitātes piesaistīto C attiecinot uz viena hektāra platību (2.1). Pieņēmumi aprēķinā:

- veģetācijas sezonas beigās (augustā) novērtētā zemsedzes veģetācijas biomasa ir vienāda ar tās ikgadējo neto ekosistēmas produktivitāti un ikgadējo atmirumu;
- ikgadējā koku sīksakņu produkcija vienāda ar divos un vienā veģetācijas sezonā cilindrā ieaugušo sīksakņu biomasas starpību (Bhuiyan et al., 2017);
- ar ieaugšanas cilindra metodi noteiktā sīksakņu produkcija ir vienāda ar ikgadējo sīksakņu atmirumu (Laiho et al., 2014b);
- viss ikgadējās vainaga nobirās, atmirušajās sīksaknēs, kā arī zemsedzes veģetācijas biomasas atmirumā esošais C ikgadēji pāriet augsnes C krātuvē.

$$C_{ienese} = \frac{m_{ienese} \bullet 10000}{S} \bullet \frac{C}{100}, \text{ kur} \quad (2.1)$$

C_{ienese} – ikgadējā augsnes C ienese ar vainaga nobirām, koku sīksaknēm vai zemsedzes veģetāciju (virszemes vai sakņu), t C ha^{-1} gadā;

m_{ienese} – gada laikā no nobiru uztvērēja ievākto nobiru biomasa, ikgadējā sīksakņu produkcija vai no paraugošanas vietas ievāktās zemsedzes veģetācijas (virszemes vai sakņu) biomasa, sausna t;

S – nobiru uztvērēja laukums, sīksakņu ieaugšanas cilindra šķērslaukums vai zemsedzes veģetācijas paraugošanas vietas laukums, m^2;
C – C koncentrācija absolūti sausās nobirās, koku sīksaknēs, zemsedzes veģetācijas virszemes vai sakņu biomasā, %.

2.5. Augsnes SEG emisiju aprēķināšana un aprēķina faktoru izstrāde

Augsnes SEG emisiju aprēķināšanai sākotnēji veikta lineārās regresijas analīze, izmantojot datus par SEG gāzu koncentrāciju kamerā uzreiz pēc kameras uzstādīšanas uz gredzena un 10; 20; 30 minūtes pēc pirmā parauga paņemšanas. Iegūto rezultātu ticamības nodrošināšanai veikta loģiskā datu kontrole, regresijas analīzē neiekļaujot datus, kas neseko lineārai gāzu koncentrācijas izmaiņai. Papildus novērtēts katras iegūtā lineārā regresijas vienādojuma determinācijas koeficients, un turpmākā augsnes SEG emisiju aprēķinā izmantoti iegūtie slīpuma koeficienti (mainīgais "b" vienādojumā 2.2) no vienādojumiem ar $R^2 > 0,7$, izņemot gadījumus, kad novērtētā maksimālās un minimālās SEG koncentrācijas starpība kamerā ir mazāka par pielietotās paraugu testēšanas ar gāzu hromatogrāfu metodes nenoteiktību. Iegūtie lineāro vienādojumu slīpuma koeficienti, kas raksturo SEG koncentrācijas izmaiņu kamerā gāzu paraugu ņemšanas laikā, izmantoti, lai ar ideālās gāzes stāvokļa vienādojumu aprēķinātu gaisa un augsnes, tajā skaitā kamerā ietvertās veģetācijas, SEG gāzu apmaiņu:

$$SEG = \frac{M \bullet P \bullet V \bullet b}{R \bullet T \bullet S} \text{, kur} \qquad (2.2)$$

SEG – SEG apmaiņa starp atmosfēru un augsni, tajā skaitā kamerā ietverto veģetāciju, µg SEG m^{-2} h^{-1};
M – SEG molmasa, g mol^{-1};
P – gaisa spiediens kamerā = 101 300 Pa;
V – kameras tilpums = 0,063 m^3;
b – lineāras regresijas vienādojuma slīpuma koeficients, kas raksturo gāzu koncentrācijas izmaiņu kamerā laika vienība, ppm h^{-1};
R – universālā gāzu konstante = 8,314 m^3 Pa K^{-1} mol^{-1};
T – gaisa temperatūra, K;
S – augsnes gredzena laukums = 0,1995 m^2.

Pieņemts, ka ar 2.2. vienādojumu novērtētā CH_4 un N_2O apmaiņa ir vienāda ar augsnes CH_4 un N_2O emisijām. Gada kopējo augsnes CH_4 un N_2O emisiju aprēķinā pieņemts, ka veiktie ikmēneša emisiju mērījumu rezultāti ir vienādi ar attiecīgā mēneša kopējām augsnes emisijām parauglaukumā. Attiecīgi ikgadējās augsnes SEG emisijas izmēģinājumu objektā aprēķinātas kā ikmēneša augsnes SEG emisiju summa:

$$SEG_{ikgadēji} = \sum SEG_{ikmēneša}\,(jan...dec) \text{, kur} \qquad (2.3)$$

$SEG_{ikgadēji}$ – ikgadējās augsnes SEG emisijas izmēģinājumu objektā, kg ha^{-1} gadā;
$SEG_{ikmēneša}$ – mēneša kopējās augsnes SEG emisijas izmēģinājumu objektā, kg ha^{-1} mēnesī.

Gada kumulatīvo augsnes kopējo CO_2 emisiju aprēķins veikts, interpolējot ikmēneša augsnes CO_2 emisiju mērījumu rezultātus, pielietojot:

- R_{10} un Q_{10} parametrus (Varik et al., 2015; Uri et al., 2017; Kriiska et al., 2019b);
- pētījumā novērtēto sakarību starp gaisa un augsnes 5 cm dziļumā temperatūrām;
- kā arī individuālam parauglaukumam tuvākās LVĢMC meteoroloģiskas stacijas datus par stundas vidējo gaisa temperatūru.

R_{10} parametra vērtība ir vienāda ar *Rkop*, kad augsnes temperatūra ir 10°C. Savukārt, Q_{10} parametrs raksturo *Rkop* izmaiņas, augsnes temperatūrai pieaugot par 10°C. R_{10} un Q_{10} parametru vērtības aprēķinātas atbilstoši katrā parauglaukumā iegūtajiem empīriskajiem datiem. Emisiju datu interpolācijas aprēķina gaitā sākotnēji noteikts eksponenciāla 2.4. vienādojuma, kas raksturo augsnes temperatūras un CO_2 emisiju sakarību katrā parauglaukumā, *b* koeficients.

$$Rkop = ae^{bTa}\text{ , kur} \tag{2.4}$$

Rkop – augsnes kopējās CO_2 emisijas, µg CO_2-C m^{-2} h^{-1};
a, *b* – eksponenciāla vienādojuma koeficienti;
Ta – augsnes temperatūra 5 cm dziļumā, °C.

Ar eksponenciālu vienādojumu (2.4) iegūtais koeficients *b* izmantots, lai aprēķinātu Q_{10} vērtību (2.5).

$$Q_{10} = e^{10b}\text{ , kur} \tag{2.5}$$

Q_{10} – augsnes elpošanas temperatūras jutīguma koeficients;
b – eksponenciāla vienādojuma koeficients.

Ar 2.4. vienādojumu novērtētā parametra R_{10} un ar 2.5. vienādojumu novērtētā parametra Q_{10} vērtība, kombinācijā ar datiem par augsnes temperatūru izmantota, lai interpolētu *Rkop* (2.6) katrā parauglaukumā. Augsnes temperatūras izmaiņas laikā noteiktas, izmantojot regresijas vienādojumu, kas raksturo augsnes un gaisa temperatūras mērījumus sakarību pētījuma parauglaukumos un datus par ik stundas vidējo gaisa temperatūru no LVĢMC meteoroloģiskajām stacijām.

$$Rkop = R_{10}\, Q_{10}^{((Ta-10)/10)}\text{ , kur} \tag{2.6}$$

Rkop – augsnes kopējās CO_2 emisijas, µg CO_2-C m^{-2} h^{-1};
Ta – augsnes temperatūra 5 cm dziļumā, °C;
R_{10} – augsnes kopējās CO_2 emisijas, tai esot 10°C temperatūrā 5 cm dziļumā, µg CO_2-C m^{-2} h^{-1};
Q_{10} – augsnes elpošanas temperatūras jutīguma koeficients.

Lai iegūto *Rkop* rezultātu pārrēķinātu uz augsnes heterotrofo elpošanu (*Rhet*), piemērots līdzšinējos pētījumos izstrādāts vienādojums (Bond-Lamberty et al., 2004). Vienādojums piemērots individuālos parauglaukumos novērtēto *Rkop* emisiju pārrēķinam.

$$\ln(Rhet) = 1{,}22 + 0{,}73\ln(Rs) \quad R^2 = 0{,}81 \quad P < 0{,}001 \text{ , kur} \qquad (2.7)$$

Rhet – augsnes heterotrofā elpošana, g C m^{-2} gadā;
Rs – augsnes kopējā elpošana bez virszemes autotrofās elpošanas, g C m^{-2} gadā.

2.6. Meža ekosistēmas SEG emisiju novērtēšana

Meža ekosistēmas SEG emisiju novērtējumā veikta SEG emisiju un CO_2 piesaistes dinamikas modelēšana atbilstoši mežaudžu attīstībai 240 gadu meža apsaimniekošanas posmā. Novērtējumā ietverti:

- pētījuma rezultāti par ikgadējām augsnes SEG emisijām un C ienesi ar koku vainagu nobirām, zemsedzes veģetācijas virszemes un sakņu biomasu, kā arī koku sīksakņu atmirumu;
- līdzšinējo pētījumu rezultāti par ikgadējo augsnes C ienesi ar sūnu un sīkkrūmu atmirumu;
- Meža resursu monitoringa (MRM) un LVMI "Silava" ilgtermiņa meža resursu prognožu modeļa (AGM) datos balstīts C piesaistes kokaugu dzīvajā un nedzīvajā biomasā novērtējums atbilstoši mežaudzes attīstībai, tajā skaitā ikgadējam koksnes pieaugumam, dabiskajam atmirumam un mežizstrādei;
- C piesaistes koksnes produktos un biokurināmā aizvietošanas efekta novērtējums atbilstoši pieņēmumiem par mežizstrādē sagatavoto apaļo kokmateriālu veidu struktūru, biokurināmā īpatsvaru;
- CH_4 emisiju no meliorācijas grāvjiem novērtējums atbilstoši Vanags-Duka et. al. (2022) ziņotajām vidējām emisijām;
- netiešo augsnes CO_2 emisiju (DOC izskalošanās) novērtējums atbilstoši IPCC noklusētajiem EF (Hiraishi et al., 2014).

Novērtējums veikts atbilstoši SEG inventarizācijas un IPCC vadlīniju pieejai SEG emisiju novērtējumā meža zemē, kas nav mainījusi zemes izmantošanas veidu vismaz 20 gadus. Attiecīgi, meža SEG emisiju dinamikas aprēķins balstīts uz ikgadējā C uzkrājuma maiņas tā krātuvēs (augsne, dzīvā koku biomasa, atmirusī koksne un koksnes produktos), kā arī augsnes CH_4 un N_2O emisiju, tajā skaitā no meliorācijas grāvjiem, novērtējumu. Ikgadējā C uzkrājuma un SEG emisiju dinamikas novērtējums veikts, pielietojot AGM datus par egles, priedes un melnalkšņa mežu augšanas gaitu un mežizstrādi platlapju kūdreņa, dumbrāja un liekņas meža tipos – mežaudzes vecuma dinamiku un augošu, atmirušu, kā arī nocirstu koku augstuma, caurmēra, skaita un krājas ikgadējiem rādītājiem meža apsaimniekošanas ciklā.

<u>Augsnes C uzkrājuma izmaiņas</u> novērtētas, summējot ikgadējos C zudumus *Rhet* rezultātā un ikgadējo C ienesi ar koku vainaga nobirām, zemsedzes veģetāciju, koku sīksaknēm, sūnām un sīkkrūmiem. *Rhet* ikgadējo augsnes C zudumu un C ieneses ar zemsegas veģetāciju un koku sīksaknēm aprēķinā piemērotas pētījuma rezultātā iegūtas fiksētas vērtības VKS un meža zemes statusu (mežaudze vai izcirtums) sadalījumā, atbilstoši pieņēmumam par apsaimniekotu mežu ikgadējo šķērslaukumu dinamiku. Par kritēriju zemes statusu iedalījumam pieņemts mežaudzes kritiskais šķērslaukuma – egles, bērza un melnalkšņa mežiem attiecīgi 6; 4 un 5 m^2 ha^{-1}. Tādējādi aprēķinā piemēroti pētījuma izcirtumos novērtētās augsnes C zudumu un

ieneses fiksētās vērtības, ja mežaudzes šķērslaukums ir mazāks par kritisko šķērslaukumu, bet pētījuma mežaudzēs iegūtās vērtības piemērotas – ja šķērslaukums lielāks par kritisko šķērslaukumu.

Augsnes ikgadējās C ieneses ar koku vainaga nobirām, sūnām un sīkkrūmiem aprēķinā izmantoti vienādojumi, kas raksturo C ienesi atkarībā no pieņēmuma par mežaudzes šķērslaukuma vai vecuma dinamiku meža apsaimniekošanas cikla laikā. C ieneses ar koku vainaga nobirām aprēķinā izmantoti pētījumā izstrādāti vienādojumi, kas raksturo ikgadējo C ienesi atkarībā no mežaudzes šķērslaukuma egļu un lapkoku mežos. Savukārt, ikgadējās C ieneses ar sūnām un sīkkrūmiem aprēķinā izmantoti līdzšinējos pētījumos izstrādāti vienādojumi (Muukkonen & Mäkipää, 2006), kas raksturo biomasu atkarībā no mežaudzes vecuma. Pieņemts, ka: sīkkrūmu un sūnu virszemes biomasas ikgadējā atmiruma īpatsvars ir attiecīgi 25% un 33% (Muukkonen & Mäkipää, 2006) ar vidējo C saturu 47,5% (FAO, 2015); 70% kopējās C oglekļa ienesi ar sīkkrūmiem un sūnām veido zemē esošā biomasa (Havas & Kubin, 1983; Mälkönen, 1974; Palviainen et al., 2005).

Meliorāciju grāvju CH_4 emisijas un augsnes netiešās CO_2 emisijas DOC izskalošanās rezultātā aprēķinātas atbilstoši IPCC noklusētajiem EF. Pieņemts, ka hidromeliorācijas grāvju platību īpatsvars ir 3% un CH4 emisijas 10,3 kg CH_4 ha^{-1} gadā (Vanags-Duka et al., 2022). Savukārt ar DOC saistītās emisijas hidromeliorētās platībās un platībās ar saglabātu hidroloģisko režīmu aprēķinātās ar EF, attiecīgi 1,1 un 0,9 t CO_2 ha^{-1} gadā un nenoteiktību 66,7% (Hiraishi et al., 2014).

Biomasas C uzkrājuma ikgadējo izmaiņu aprēķina pamatā ir individuālu koku biomasas aprēķina alometriskie vienādojumi (Liepiņš et al., 2018) un AGM dati par dzīvo, nedzīvo un nocirsto koku rādītāju dinamiku meža apsaimniekošanas ciklā, kā arī kā arī vidējo svērto C saturu koku biomasā biomasā (Bārdule et al., 2021c). Ikgadējā C uzkrājuma izmaiņa dzīvajā biomasā noteikta, aprēķinot starpību starp C uzkrājumu aprēķina un iepriekšējā gadā, neskaitot C ikgadēji nocirsto un atmirušo koku biomasā. Ikgadēji nocirsto koku sakņu un koku vainaga biomasas C maina krātuvi uz nedzīvo biomasu, bet stumbra masas C dati tiek izmantoti C aprites aprēķināšanai nocirstas koksnes produktos. Savukārt, ikgadēji atmirušo koku gadījumā – viss saistītais C maina krātuvi uz atmirušo koksni. Aprēķinā pieņemts, ka nedzīvās koksnes C krātuvē uzkrātais C pāriet atmosfērā 20 gadu laikā. Lai izvairītos no novirzes aprēķinā, pieņemts, ka 240 gadu meža apsaimniekošanas cikla sākumā C uzkrājums atmirušajā koksnē atbilst meža resursu MRM datiem par vidējo C uzkrājumu. Attiecīgi ikgadējā C uzkrājuma izmaiņa nedzīvajā koksnē aprēķināta no aprēķina gadā atmirušās koksnes C un iepriekšējā gadā esošā C uzkrājuma atmirušajā koksnē summas atņemot proporciju (5%) no C, kas aprēķina gadā pāriet atmosfērā.

Ikgadējās C uzkrājuma izmaiņas koksnes produktos pamatā ir pieņēmums par mežizstrādē sagatavoto vidējo kokmateriālu veidu struktūru Pieņemts arī, ka 50% zāģmateriālu un papīrmalkā esošā C koksnes produktu krātuvē nenonāk, ražošanas procesā radušo zudumu ietekmē. Ikgadējā C izmaiņa aprēķināta kā C uzkrājuma starpība aprēķina gada beigās un sākumā jeb iepriekšējā gada beigās atbilstoši Nacionālajā SEG inventariācijā pielietotajai metodikai (Skrebele et al., 2021). Analoģiski kā gadījumā ar C uzkrājumu atmirušajā koksnē, pieņemts, ka 240 gadu meža apsaimniekošanas periodā sākumā koksnes produktos uzkrātais C vienāds ar vidējo C uzkrājumu koksnes produktos atbilstoši SEG inventarizācijā (Skrebele et al., 2021) ziņotajiem rezultātiem.

Biokurināmā aizvietošanas efekta aprēķins balstīts uz pieņēmumu, ka ikgadēji koksnes produktu C krātuves kategoriju pametošā koksnes daļa, koksnes produktu ražošanas zudumu koksne, kā arī nocirsto koku stumbru daļa kategorijā biokurināmais tiek izmantots kā kurināmais, kas aizvieto enerģētiskajā vērtībā ekvivalentu nesadedzinātās dabasgāzes daudzumu. Tādējādi, atbilstoši pieņēmumiem par kurināmo enerģētisko vērtību un pret to attiecināmo SEG emisiju daudzumu atkarībā no kurināmā veida (Eggleston et al., 2006), novērtēts dabasgāzes aizvietošanas ar biokurināmo efekts uz atmosfērā nenonākošām SEG.

2.7. Datu matemātiskā apstrāde

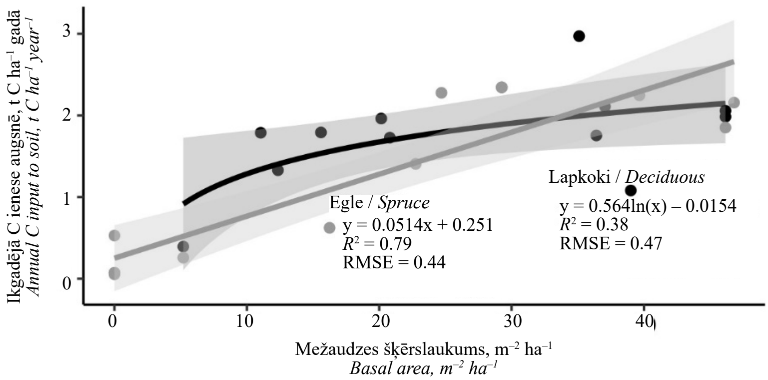
Augsnes C ieneses vai SEG emisiju mērījumu rezultātu sakarības ar ietekmējošiem faktoriem novērtētas ar regresijas analīzi, savukārt, sakarību ciešums – ar korelācijas analīzi, nosakot Pīrsona (*r*) un Spīrmena korelācijas koeficientu (ρ). Regresijas vienādojumu kvalitātes raksturošanai izmantots determinācijas koeficients (R^2) un vidējā kvadrātiskā kļūda (RMSE). Ekosistēmas vai atsevišķu C krātuvju SEG emisiju un CO_2 piesaistes līdzsvars izteikts CO_2 ekvivalentos, CH_4 un N_2O emisijas pārrēķinot ar globālās sasilšanas ietekmes potenciāla koeficientiem attiecīgi 25 un 298 (Eggleston et al., 2006). Pētījuma rezultātu nenoteiktība izteikta ar ticamības intervālu pie būtiskuma līmeņa 0,05. Kombinētu pētījumu rezultātu nenoteiktības raksturošanai novērtēta apvienotā svērtā nenoteiktība.

Datu statistiskā analīze veikta, izmantojot datorprogrammu R Studio, pie būtiskuma līmeņa $p < 0,05$. Datu izkliedes atbilstības normālajam sadalījumam pārbaudē pielietots Kalmogorova-Smirnova tests. Augsnes SEG emisiju mērījumu rezultātu vidējo vērtību salīdzināšanai izmantota neparametriskā metode Manna-Vitneja U kritērijs. Lai ņemtu vērā ietekmējošo pazīmju grupu ietekmi uz mainīgo, sakarība izteikta, veicot lineāro jauktu efektu regresijas analīzi. Lineārās regresijas vienādojumi salīdzināti ar testu ANCOVA. Dati ar ekstrēmu vērtību atlasīti, kā kritēriju izmantojot starpkvartiļu diapazonu, jeb pirmās un trešās datu kvartiles starpību (Morillas et al., 2012). Tādējādi, nodrošinot saskaņotību ar datu izkliedes vizuālo atspoguļojumu vērtībamplitūdas diagrammās, kurās atspoguļotu paraugkopas datu minimālā vērtība, pirmā kvartile, vidējais aritmētiskais (punkts), mediāna (horizontāla līnija), trešā kvartile, maksimālā vērtība un ekstrēmās vērtības, savukārt statistski būtiskas vai nebūtiskas atšķirības norādītas ar burtiem, piemēram, “a”, “b”, “c”. Datu grafiki veidoti, izmantojot datorprogrammas R pakotni ggplot2, intervāls (iekrāsojums) ap regresijas taisni vai līkni norāda tās 95% ticamības intervālu. Sakarības starp augsnes SEG emisijām un ietekmējošiem faktoriem atspoguļota ar principālo komponentu analīzes rezultātiem. Stabiņu diagrammās un tabulās nenoteiktība uzrādīta ar ticamības intervālu pie būtiskuma līmeņa 0,05.

3. REZULTĀTI UN DISKUSIJA

3.1. Oglekļa ienese augsne ar vainaga nobirām, zemsedzes veģetāciju un koku sīksaknēm

Vainaga nobiras. No vērtētajiem kokaudzes raksturlielumiem, vecumam ir ciešākā (r = 0,8) sakarība ar koku vainaga nobiru kopējo biomasu. Savukārt, šķērslaukums vislabāk spēj prognozēt gada kopējo nobiru biomasu atkarībā no koku sugas, tomēr nav novērojama tā ietekme uz ikmēneša nobiru biomasas variāciju un tendencēm. Ikmēneša nobiru biomasas variācija lapkoku un egļu audzēs ir attiecīgi vidēji 120 ± 20% un 71 ± 21%. Noteiktais C saturs nobirās (vidēji 52,1 ± 0,2%), kā arī parauglaukumos novērotā sakarība starp mežaudzes šķērslaukumu un ikgadējo nobiru biomasu nosaka, ka egļu audžu šķērslaukumam palielinoties līdz 40 m^2 ha^{-1}, ikgadējā augsnes C ienese ar nobirām lineāri palielinās līdz vidēji 2,31 t C ha^{-1} gadā. Pētījuma dati norāda, ka, lapkoku mežaudžu šķērslaukumam palielinoties līdz 10 m^2 ha^{-1}, ikgadējo nobiru biomasa strauji pieaug līdz vidēji 1,28 t C ha^{-1} gadā. Šķērslaukuma turpinot palielināties, salīdzinot ar skujkoku mežiem, lapkoku mežaudzēs ikgadējo vainaga nobiru biomasa pieaug lēnāk un tiecas stabilizēties. Lapkoku mežaudzēs ar šķērslaukumu no 11 līdz 46 m^2 ha^{-1} pētījuma periodā vidējā ikgadējā augsnes C ienese ar nobirām bija 1,86 ± 0,46 t C ha^{-1} gadā (3.1. att.).



3.1. att. **C ienese augsnē ar vainaga nobirām atkarībā no šķērslaukuma**
Fig. 3.1. ***Annual C input to soil by foliar litter as a function of basal area***

Līdzīgas nobiru ikgadējās biomasas un šķērslaukuma sakarības tendences novērotas arī citā Latvijā veiktā pētījumā mežaudzēs ar organisko augsni – lineārs nobiru biomasas pieaugums skujkoku audzēs visā šķērslaukumu diapazonā, bērzu audzēs pieņemts, ka, sasniedzot šķērslaukumu 34 m^2 ha^{-1}, nobiru biomasas apjoms ir nemainīgs (Bārdule et al., 2021d). Tomēr iepriekš veiktajā pētījumā novērtēts straujāks ikgadējais augsnes C ieneses ar nobirām pieaugums, kas, mežaudzes šķērslaukumam palielinoties līdz 40 m^2 ha^{-1}, sasniedz 2,66 t C ha^{-1} gadā un aptuveni

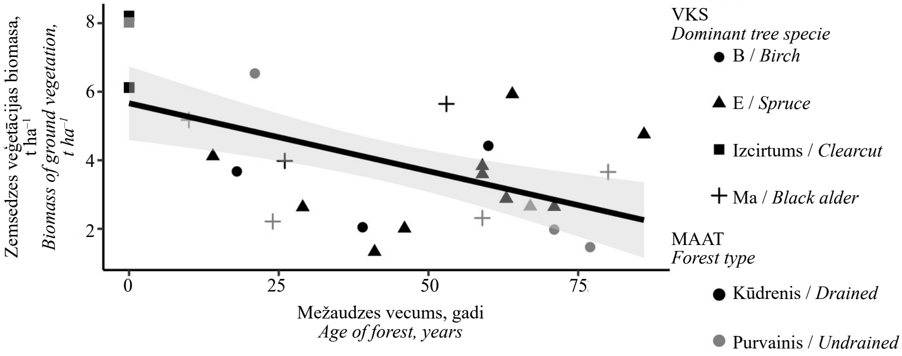
3,0 t C ha^{-1} gadā, attiecīgi bērzu un egļu audzēs. Atšķirības var būt skaidrojamas ar ikgadējo nobiru variāciju. Lai identificētu ikgadējās augsnes C ienesi ar vainaga nobirām apjoma un tā variācijas ietekmējošos faktorus, nepieciešami ilgtermiņa novērojami, kas ļautu izvērtēt sakarības starp meteoroloģiskiem apstākļiem un nobiru kopējās biomasas un tās frakciju ar dažādu C saturu sadalījuma datiem. Novērots, ka, piemēram, priežu meža skuju nobiru ikgadējā variācija var būt līdz 40% (Kouki & Hokkanen, 1992). Arī ilgtermiņa pētījumā Latvijā novērota ievērojama ikgadējā vainaga nobiru kopējā biomasa robežās no 2198 līdz 6085 kg ha^{-1} gadā (Bārdule et al., 2021a). Šādas variācijas iemesls var būt ikgadējo meteoroloģisko apstākļu dažādība un ekstremāli laikapstākļu notikumi kā vētras, kas var ievērojami ietekmēt nobiru dinamiku (Sanford et al., 1991).

Zemsedzes veģetācija. Pētījuma objektos novērtētā zemsedzes veģetācijas sakņu (pZV) biomasa (C saturs vidēji 49,7 ± 7,8%) ir robežās no 0,63 līdz 3,54 t ha^{-1} (vidēji 1,96 ± 0,30 t ha^{-1}). Izcirtumos pZV biomasa (vidēji 2,24 ± 0,96 t ha^{-1}) tiecas būt lielākā nekā mežaudzēs (vidēji 1,91 ± 0,55 t ha^{-1}), bet VKS, meža zemes statusa (mežaudze vai izcirtums) un MAAT būtiska ietekme uz vidējo pZV biomasu nav novērota.

Vidējā novērtētā virszemes zemsedzes veģetācijas (vZV) biomasa (C saturs vidēji 47,4 ± 7,2%) izcirtumu parauglaukumos 4,67 ± 0,50 t ha^{-1} (no 4,27 līdz 5,49 t ha^{-1}) veģetācijas sezonas beigās bija būtiski lielāka nekā mežaudzēs – vidēji 1,57 ± 0,30 t ha^{-1} (no 0,39 līdz 3,82 t ha^{-1}). Iegūtais rezultāts par vZV mežaudzēs ir līdzīgs tam, kas aprēķināms, izmantojot Somijas apstākļiem izstrādātu virszemes zemsedzes veģetācijas aprēķina vienādojumus, kas ar vidējo kvadrātisko kļūdu 13,6% nosaka, ka 80 gadu vecumā zemsedzes virszemes veģetācijas biomasa mežos ar kūdras augsni ir 1,65 t ha^{-1}. Promocijas darba pētījums norāda, ka izcirtumos vZV biomasa var būt aptuveni trīs reizes lielāka nekā mežos. Vidējā novērtētā vZV biomasa purvaiņa izcirtuma parauglaukumā ir 5,08 t ha^{-1}, bet kūdreņu izcirtumu parauglaukumos – vidēji 4,57 ± 0,60 t ha^{-1}.

Novērtētā zemsedzes veģetācijas (ZV) kopējā biomasa (pZV un vZV vidējais svērtais C saturs 48,2 ± 0,3%) kūdreņos ir no 1,33 t ha^{-1} līdz 5,93 t ha^{-1} (vidēji 3,48 ± 0,60 t ha^{-1}), bet purvaiņos – no 1,46 līdz 6,53 t ha^{-1} (vidēji 3,46 ± 0,66 t ha^{-1}), kopā vidēji 3,47 ± 0,66 t ha^{-1}. Izcirtumu gadījumā vidējā ZV biomasa purvainī (8,02 ± 1,63 t ha^{-1}) ir virs novērtētās vidējās biomasas kūdreņu izcirtumos (6,65 ± 1,02 t C ha^{-1}) ticamības intervāla maksimālās vērtības. Aprēķinā izmantojot Somijas apstākļiem izstrādātos biomasas vienādojumus, kā arī ņemot vērā mežu vecumstruktūru, Latvijā novērtētā ikgadējā augsnes C ienese ar ZV biomasu ir no 0,34 ± 0,01 t C ha^{-1} gadā bērza mežos un 1,29 ± 0,20 t C ha^{-1} gadā priežu mežos (Bārdule et al., 2021d).

Empīriskie dati norāda uz pozitīvu korelāciju starp ZV un augsnes auglības rādītājiem. No vērtētajiem mežaudzes parametriem, vecumam ir ciešākā korelācija (r = –0,58; p < 0,05) ar ZV biomasu. Šī sakarība ar vidējo kvadrātisko kļūdu ± 1,49 t ha^{-1} nosaka (3.2. att.), ka izcirtumos ZV biomasa ir vidēji 5,66 t ha^{-1}, kas samazinās līdz vidēji 2,46 t ha^{-1}, mežaudzei attīstoties līdz 80 gadu vecumam.

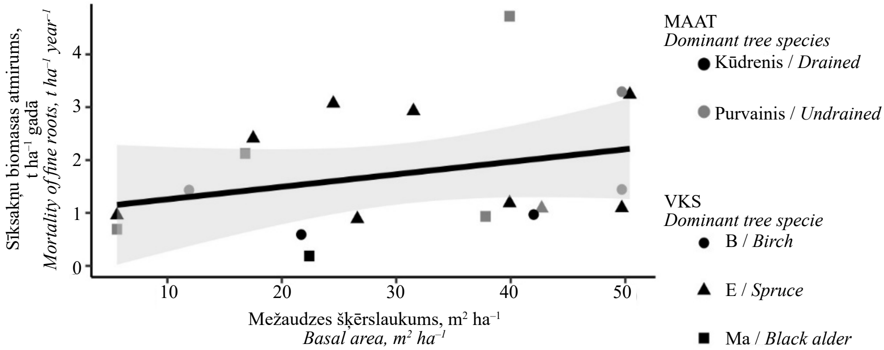


3.2. att. **Zemsedzes veģetācijas kopējā biomasa un mežaudzes vecums**
Fig. 3.2. ***Relationship between ground vegetation biomass and stand age***

<u>Koku sīksaknes</u>. Kūdreņos ar VKS B un Ma novērtētais koku sīksakņu ikgadējais atmirums ir no, attiecīgi, 0,19 ± 0,05 līdz 0,98 ± 0,87 t ha^{-1} gadā (vidēji 0,58 ± 0,44 t ha^{-1} gadā) un no 0,89 ± 0,75 līdz 3,24 ± 2,46 t ha^{-1} gadā kūdreņos ar VKS E (vidēji 1,97 ± 0,72 t ha^{-1} gadā). Savukārt lapkoku purvaiņos novērtētais sīksakņu ikgadējais atmirums ir no 0,69 ± 0,37 līdz 4,72 ± 1,15 t ha^{-1} gadā (vidēji 2,09 ± 1,07 t ha^{-1} gadā), bet E purvainī 1,09 ± 0,08 t ha^{-1} gadā.

Pētījumā novērtētais ikgadējais vidējais sīksakņu atmirums egles (1,87 ± 0,66 t ha^{-1} gadā) un lapkoku mežos (1,64 ± 0,86 t ha^{-1} gadā) iekļaujas līdzšinējo pētījumu rezultātu nenoteiktības diapazonā – ziņots, ka ikgadējā sīksakņu produkcija Ziemeļeiropas skujkoku mežos ir vidēji 2,84 ± 1,52 t ha^{-1} gadā, bet lapkoku mežos – vidēji 1,99 ± 1,01 t ha^{-1} gadā (Neumann et al., 2020). Tomēr Igaunijā veiktā pētījumā novērtētā sīksakņu ikgadējā produkcija bērza mežos ar auglīgu organisko augsni no 1,81 līdz 3,02 t ha^{-1} gadā (Uri et al., 2017) tiecas būt lielāka nekā promocijas darba pētījuma ietvaros novērtēts bērzu kūdreņos (0,59 līdz 0,97 t ha^{-1} gadā), bet vairāk atbilst novērtētajam sīksakņu produkcijas diapazonam purvaiņos (1,43 līdz 3,29 t ha^{-1} gadā). Tas var būt skaidrojams ar empīrisko datu nenoteiktības ietekmi. Promocijas darba pētījumā novērtētās sīksakņu produkcijas vidējās vērtības atkarībā no VKS un MAAT nenoteiktība ir no 30 līdz 161% (vidēji 71%). Cits iemesls var būt atšķirīgi augšanas apstākļi pētījumu objektos, jo empīriskie dati norāda, ka ikgadējā sīksakņu produkcija tiecas būt lielāka, samazinoties vidējam GŪ līmenim un augsnes auglības rādītājiem. Lielāka sīksakņu produkcija mazāk auglīgās augsnes novērota arī citos pētījumos (Leppälammi-Kujansuu et al., 2014; Lehtonen et al., 2016; Mäkelä et al., 2016; Kriiska et al., 2019a). Paaugstinātu sīksakņu produkciju purvaiņos varēja veicināt ikmēneša GŪ līmeņa dinamika. Vidējais GŪ līmenis purvaiņos vasaras mēnešos, kad sīksakņu pieaugums var būt vislielākais (Varik et al., 2015), bija dziļāks par 40 cm, kā tas ir bijis arī minētajā Igaunijas pētījumā ar hidromeliorētu augsni.

No vērtētajiem mežaudzes raksturlielumiem, šķērslaukumam ir ciešākā sakarība ar ikgadējo koku sīksakņu atmirumu (r = 0,30). Iegūtie dati norāda, ka, mežaudzes šķērslaukumam pieaugot no 10 līdz 40 m^2 ha^{-1}, ikgadējais sīksakņu biomasas atmirums palielinās no vidēji 0,64 līdz 0,99 t ha^{-1} gadā (3.3. att.). Tomēr no pētījumā iegūtajiem empīriskajiem datiem izveidojamā regresijas vienādojuma ikgadējā sīksakņu atmiruma prognozei ir liela vidējā kvadrātiskā kļūda ± 1,43 t ha^{-1} gadā, jeb 81% no izmēģinājumu objektos novērtētās vidējās ikgadējā sīksakņu atmiruma vērtības.



3.3. att. **Sakarība starp sīksakņu atmirumu un mežaudzes šķērslaukumu**
Fig. 3.3. ***Relationship between fine root biomass and stand basal area***

3.2. Augsnes SEG emisiju aprēķina faktori un vienādojumi

Augsnes CH_4 emisijas. Kūdreņos veikto ikmēneša CH_4 mērījumu rezultātu vidējais variācijas koeficients ir 60%, bet purvaiņos tas ir 268%. Kūdreņos augsnes CH_4 emisiju ikmēneša mērījumu rezultātu vidējā vērtība pētījuma objektos ir robežās no –7,15 ± 2,86 līdz 2,87 ± 14,04 kg CH_4-C ha^{-1} gadā, bet purvaiņos – robežās no –4,56 ± 2,35 līdz 497,15 ± 1558,67 kg CH_4-C ha^{-1} gadā.

Augsnes CH_4 emisiju ikmēneša mērījumiem raksturīga liela variācija. Kā ekstrēmi identificēti augsnes ikmēneša CH_4 emisiju mērījumi < –12,26 un > 5,61 kg CH_4-C ha^{-1} gadā. Ekstrēmas vērtības novērotas trīs purvaiņu objektos (ekstrēmo emisiju vidējā vērtība 877,76 ± 1424,652 kg CH_4-C ha^{-1} gadā) un četros kūdreņu objektos (ekstrēmo emisiju vidējā vērtība 27,53 ± 23,48 kg CH_4-C ha^{-1} gadā). Izteikti lielas ekstrēmo emisiju vērtības (vidēji 1355,81 ± 1682,84 kg CH_4-C ha^{-1} gadā), deviņās parauglaukuma apsekošanas reizēs sasniedzot līdz 4933,09 ± 25517,45 kg CH_4-C ha^{-1} gadā, noteiktas vienā no purvaiņu parauglaukumiem ar VKS melnalksnis. Līdzīgs novērojums konstatēts pētījumā ziemeļu reģiona kūdrājos, kurā novērots, ka trīs ekstrēmi lielu CH_4 emisiju epizodēs no augsnes atmosfērā nonāca 1020 kg CH_4-C ha^{-1} (Glaser et al., 2004). Pārējos deviņos purvaiņu objektos ekstrēmo emisiju vidējā vērtība 17,27 ± 9,3 kg CH_4-C ha^{-1} gadā ir mazāka par noteikto ekstrēmo emisiju vidējo vērtību kūdreņu objektos. Attiecīgi pētījuma rezultāti norāda, ka izteikti ekstrēmas augsnes CH_4 emisijas, kas var būtiski ietekmēt noteikto vidējo emisiju daudzumu, var būt sagaidāmas aptuveni 10% no purvaiņu platībām. Turklāt lielāku CH_4 emisiju tendence novērojama pētījuma mežaudzēs ar valdošo koku sugu melnalksnis. Augsnes CH_4 emisiju telpiskā neviendabība novērtēta arī citā pētījumā – ziemeļu reģiona kūdrājos 10% platības ar GŪ līmeni tuvu augsnes virsmai (augsnes piesātinājuma apstākļos) var radīt līdz pat 45% no kopējām CH_4 emisijām (Sachs et al., 2011).

Pētījumā novērtētās ikgadējās augsnes CH_4 emisijas kūdreņos ir no –8,2 līdz 15,3 kg CH_4-C ha^{-1} gadā (vidēji –3,47 ± 0,94 kg CH_4-C ha^{-1} gadā), bet purvaiņos no –6,5 līdz 1016,2 kg CH_4-C ha^{-1} gadā (vidēji 106,6 ± 101,0 kg CH_4-C ha^{-1} gadā). Novērtētās vidējās ikgadējās augsnes CH_4 emisijas melnalkšņa kūdreņos un purvaiņos ir attiecīgi 6,8 ± 16,6 un 199,8 ± 393,2 kg CH_4-C ha^{-1} gadā (3.1. tab.). Mežaudzēs ar VKS bērzs un egle novērtētas negatīvas ikgadējās augsnes CH_4 emisijas (vidēji –4,4 ± 1,2 kg CH_4-C ha^{-1} gadā) gan kūdreņos, gan purvaiņos. Viennozīmīga MAAT ietekme uz tās CH_4 emisijām izcirtumā nav novērota. Novērtētās ikgadējās augsnes CH_4 emisijas izcirtumu pētījuma objektos ir no –6,0 līdz 6,88 kg CH_4-C ha^{-1} gadā (vidēji –2,4 ± 4,6 kg CH_4-C ha^{-1} gadā). Emisiju diapazons kūdreņos ir līdzīgs kāds atkarībā no augsnes auglības novērtēts Somijā – no –2,8 kg CH_4-C ha^{-1} gadā mazauglīgās augsnēs, līdz 11,6 kg CH_4-C ha^{-1} gadā auglīgās augsnēs (Ojanen et al., 2013). Savukārt novērtētās vidējās ikgadējās purvaiņu augsnes CH_4 emisijas iekļaujas IPCC vadlīniju noklusētā CH_4 no auglīgas organiskās augsnes ar atjaunotu hidroloģisko režīmu boreālā zonā EF 95% ticamības intervālā no 0 līdz 493 kg CH_4-C ha^{-1} gadā (vidēji 137 kg CH_4-C ha^{-1} gadā) (Hiraishi et al., 2014).

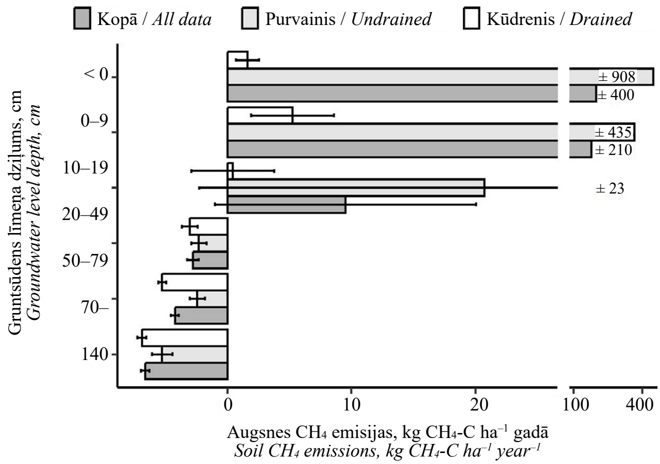
3.1. tabula / *Table 3.1*

Ikgadējās augsnes CH_4 emisijas izmēģinājumu objektos
Annual soil CH_4 emissions at study sites

Valdošā koku suga *Dominant tree specie*	Kūdrenis *Drained sites*	Purvainis *Undrained sites*
	kg CH_4-C ha^{-1} gadā / *kg CH_4-C ha^{-1} $year^{-1}$*	
Bērzs / *Birch*	–1.7 ± 2.0	–3.7 ± 2.8
Egle / *Spruce*	–5.5 ± 1.0	–2.4 ± 1.2
Izcirtums / *Clearcut*	–4.7 ± 1.0	6.9 ± 6.2
Melnalksnis / *Black alder*	6.8 ± 16.6	199.8 ± 393.2
Melnalksnis [1] / *Black alder* [1]	-	–0.9 ± 0.4
Melnalksnis [2] / *Black alder* [2]	-	1016.20 ± 883.3
Vidēji / *Mean*	–3.5 ± 0.9	100.6 ± 101.0

[1] izņemot parauglaukumu ar ekstrēmām emisijām / *excluding study site with outlier emissions*;
[2] parauglaukums ar ekstrēmām emisijām / *site with outlier emissions*.

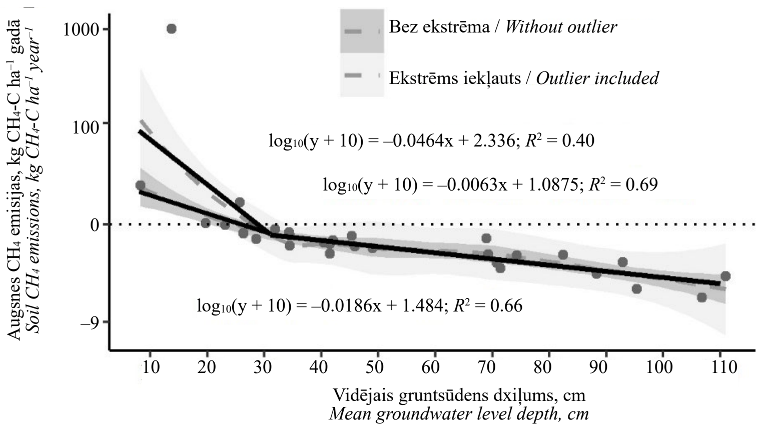
Lai gan ikmēneša GŪ līmeņa un CH_4 emisiju mērījumu rezultātiem ir cieša korelācija ($r = 0,8$), emisiju mērījumu nenoteiktība nav vienmērīga visā GŪ līmeņa dziļumu diapazonā. Novērtējot vidējās augsnes CH_4 emisijas GŪ līmeņa gradācijas klasēs, uzskatāmi redzams, ka emisiju nenoteiktība ir ievērojami lielāka, GŪ līmenim esot seklāk par 20 cm. GŪ līmenim esot diapazonā no augsnes virskārtas līdz 20 cm dziļumam, pētījumā novērtētās vidējās CH_4 emisijas ir 87,5 ± 97,3 kg CH_4-C ha^{-1} gadā, savukārt, gruntsūdenim esot dziļākam, mērījumu vidējā vērtība ir –4,4 ± 0,2 kg CH_4-C ha^{-1} gadā (3.4. att.). Arī līdzšinējo pētījumu, kas veikti mērenās un boreālās zonas kūdrājos un purvos, rezultāti norāda, ka būtiskas CH_4 emisijas ir sagaidāmas, GŪ līmeņa dziļumam esot seklāk par 20 cm (Couwenberg & Fritz, 2012). Ņemot vērā datu nenoteiktību, promocijas darba un līdzšinējos pētījumos iegūtie rezultāti ir salīdzināmi – boreālā zonā novērtētas CH_4 emisijas no kūdrājiem, gruntsūdenim esot seklāk un dziļāk par 20 cm no augsnes virskārtas, ir attiecīgi no –1,7 līdz 525 kg CH_4 ha^{-1} gadā (vidēji 56 kg CH_4 ha^{-1} gadā) un no –1,1 līdz 51 kg CH_4 ha^{-1} gadā (vidēji 8,6 kg CH_4-C ha^{-1} gadā) (Couwenberg & Fritz, 2012). Šie novērojumi promocijas darba un līdzšinējos pētījumos skaidrojami ar to, ka 20 cm augsnes slānis ar aerobiem apstākļiem ir pietiekams, lai oksidētu visu vai lielāko daļu no augsnes slānī ar anaerobiem apstākļiem radītā CH_4 pirms tas nonāk atmosfērā (Hornibrook et al., 2009).



3.4. att. **Augsnes CH_4 emisijas atkarībā no gruntsūdens līmeņa dziļuma gradācijas klasēm**

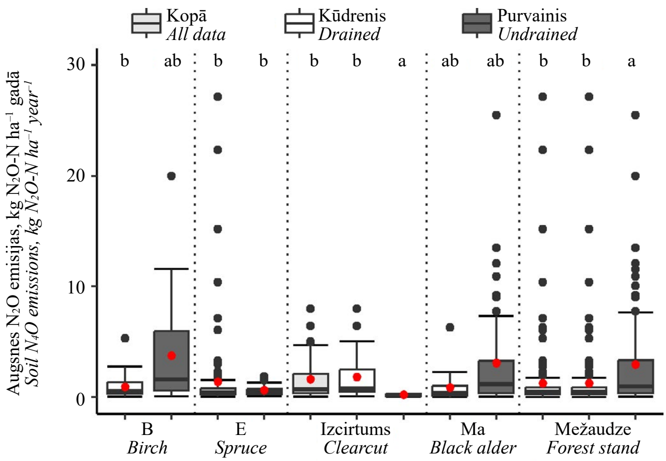
Fig. 3.4. ***Soil CH_4 emissions depending on gradation classes of groundwater level depth***

GŪ līmeņa mērījumu pētījumu objektos vidējā vērtībai ir vidēji cieša (r = –0,64) un cieša (r = –0,88) korelācija ar aprēķinātajām ikgadējām kopējām augsnes CH_4 emisijām, attiecīgi ņemot un neņemot vērā pētījuma objektu ar ekstrēmu gada kumulatīvo emisiju vērtību. Attiecīgi vidējam GŪ līmenim un gada kumulatīvajām emisijām ir tik pat cieša korelācija kā savstarpēji ikmēneša GŪ un CH_4 emisiju mērījumu rezultātiem. GŪ līmenis nosaka augsnes slāņu ar aerobiem un anaerobiem apstākļiem biezumu, attiecīgi arī CH_4 producējošo vai patērējošo mikroorganismu dažādību un proporciju, kas regulē līdzsvaru starp augsnes CH_4 emisijām un tā oksidēšanu augsnē (Couwenberg & Fritz, 2012). Vidējais GŪ līmenis var precīzi norādīt uz ikgadējām augsnes CH_4 emisijām, jo esošie metanogēnie un metanofīlie mikroorganismi ir labi pielāgojušies nelabvēlīgo apstākļu stresam un saglabājas bagātīgā daudzumā noteiktā dziļumā zem augsnes virsmas neatkarīgi no GŪ līmeņa svārstībām (Kettunen et al., 1999; Knorr & Blodau, 2009; Kip et al., 2012). Atbilstoši pētījumā iegūtajiem gada kumulatīvo emisiju un vidējā GŪ līmeņa dziļuma rezultātiem, izdalāmi divi GŪ līmeņa dziļuma diapazoni ar robežvērtību 31 cm. GŪ līmenim esot dziļāk par 31 cm, lineāras regresijas grafika taisnes pārklājas neatkarīgi no tā vai analīzē tiek ņemts vērā objekts ar statistiski ekstrēmu ikgadējo emisiju vērtību. GŪ līmenim esot seklāk par 31 cm, ekstrēmā ikgadējo CH_4 emisiju vērtība (1036,7 ± 834,4 kg CH_4-C ha^{-1} gadā) ievērojami ietekmē lineāras regresijas vienādojuma slīpuma koeficientu (3.5. att.). Šie diapazoni atbilst IPCC definētajai robežvērtībai 30 cm, kas nodala sekli vai dziļi drenētas augsnes (Hiraishi et al., 2014).



3.5. att. **Ikgadējās augsnes CH_4 emisijas atkarībā no vidējā GŪ līmeņa**
Fig. 3.5. ***Annual soil CH_4 emissions based on mean groundwater level***

Augsnes N_2O emisijas. Kūdreņos augsnes N_2O mērījumu vidējais rezultāts ir no 0,6 ± 0,6 kg N_2O-N ha^{-1} gadā pētījuma mežaudzēs ar VKS melnalksnis līdz 1,5 ± 1,3 kg N_2O-N ha^{-1} gadā izcirtumos. Savukārt, purvaiņos noteiktās augsnes vidējās N_2O emisijas ir no 0,0 ± 0,1 līdz 3,3 ± 4,0 kg N_2O-N ha^{-1} gadā attiecīgi izcirtuma parauglaukumā un melnalkšņa mežaudzēs. Bērza un melnalkšņa mežaudzēs, kā arī izcirtumos augsnes hidroloģiskam režīmam ir būtiska ietekme uz augsnes N_2O mērījumu vidējo rezultātu, savukārt egļu purvaiņos un kūdreņos emisiju mērījumu vidējā vērtība būtiski neatšķiras. Kūdreņu augsnes emisiju mērījumu mežaudzēs ar valdošo kokus sugu bērzs, melnalksnis un egle vidējās vērtības attiecīgi 0,842 ± 0,33; 0,615 ± 0,54 un 1,092 ± 0,60 kg N_2O-N ha^{-1} gadā nav būtiski atšķirīgas. Purvaiņos situācija ir pretēja, emisiju mērījumu rezultātu vidējās vērtības bērza (2,85 ± 1,46 kg N_2O-N ha^{-1} gadā), egles (0,64 ± 0,33 kg N_2O-N ha^{-1} gadā) un melnalkšņa (3,31 ± 1,52 kg N_2O-N ha^{-1} gadā) mežaudzēs ir savstarpēji būtiski atšķirīgas. Augsnes N_2O mērījumu vidējās vērtības purvaiņos un kūdreņos, attiecīgi, 2,6 ± 0,9 un 1,1 ± 0,4 kg N_2O-N ha^{-1} gadā ir būtiski (p = 0,01) atšķirīgas (3.6. att.).



3.6. att. **Augsnes N_2O emisiju mērījumu rezultātu izkliede**

Fig. 3.6. ***Variation of soil N_2O emission measurement results***

Ja netiek ņemtas vērā augsnes N_2O emisiju mērījumu ekstrēmās vērtības, *Ta* izmaiņas izskaidro 44% emisiju variācijas. Kūdreņu augsnes *Ta* mērījumu rezultātiem ir vidēji cieša korelācija (r = 0,48) ar emisiju mērījumiem, bet purvaiņu augsnes *Ta* mērījumiem – ļoti vāja. Ja korelācijas analīzē izmanto visu empīrisko materiālu, gan augsnes temperatūras, gan GŪ līmeņa mērījumu rezultātiem ir vāja korelācija (r = 0,3) ar augsnes N_2O mērījumu rezultātiem.

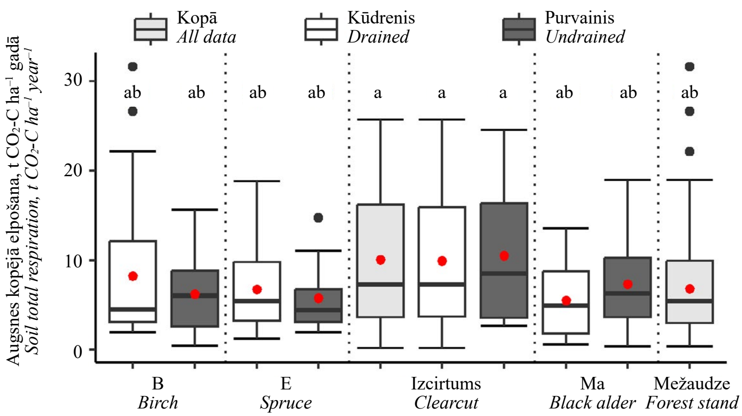
Pētījumā novērtētās ikgadējās kūdreņu augsnes N_2O emisijas vidēji 1,1 ± 0,4 kg N_2O-N ha^{-1} gadā ir mazākas par IPCC noklusēto hidromeliorētas auglīgas organiskās augsnes boreālā zonā EF 3,2 kg N_2O-N ha^{-1} gadā (95% ticamības intervāls no 1,9 līdz 4,5 kg N_2O-N ha^{-1} gadā), bet iekļaujas mērenās joslas EF (2,8 kg N_2O-N ha^{-1} gadā) 95% ticamības intervālā no –0,57 līdz 6,1 kg N_2O-N ha^{-1} gadā (Hiraishi et al., 2014). Somijā novērtētās hidromeliorētas organiskās augsnes atkarībā no tās auglības ikgadējās N_2O emisijas ir robežās no 0,18 ± 0,04 līdz 2,11 ± 0,64 kg N_2O-N ha^{-1} gadā (Statistics Finland, 2014). IPCC vadlīnijās pieņemts, ka organiskās augsnes ar atjaunotu hidroloģisko režīmu N_2O emisijas ir nebūtiskas, bet šajā pētījumā novērtētās vidējās ikgadējās purvaiņu augsnes emisijas (2,6 ± 0,9 kg N_2O-N ha^{-1} gadā) ir lielākas nekā no kūdreņu augsnes (3.2. tab.).

3.2. tabula / *Table 3.2*

Ikgadējās augsnes N_2O emisijas izmēģinājumu objektos
Annual soil N_2O emissions at study sites

Valdošā koku suga *Dominant tree specie*	Kūdrenis / *Drained sites*	Purvainis / *Undrained sites*
	kg N_2-N ha^{-1} gadā / *kg N_2-N ha^{-1} year^{-1}*	
Bērzs / *Birch*	0.9 ± 0.6	2.7 ± 3.1
Egle / *Spruce*	1.0 ± 0.9	0.6 ± 0.3
Izcirtums / *Clearcut*	1.5 ± 1.3	0.0 ± 0.1
Melnalksnis / *Black alder*	0.6 ± 0.6	3.3 ± 4.0
Vidēji / *Mean*	1.1 ± 0.4	2.6 ± 0.9

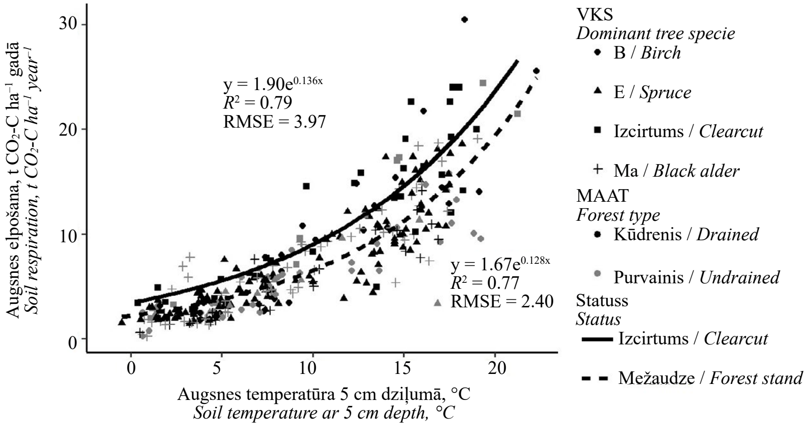
Augsnes elpošanas CO_2 emisijas. *Rkop* mērījumu vidējās vērtības pētījuma mežaudzēs ar dažādu VKS savstarpēji nav būtiski atšķirīgas ($p > 0,05$). *Rkop* mērījumu vidējās vērtības būtiski neatšķiras arī meža tipu ($p > 0,05$) vai MAAT ($p = 0,34$) sadalījumā (3.7. att.). Attiecīgi arī vidējā novērtētā *Rkop* kūdreņos (7,35 ± 0,89 t CO_2-C ha^{-1} gadā) un purvaiņos (7,02 ± 0,96 t CO_2-C ha^{-1} gadā) būtiski neatšķiras ($p = 0,34$). Lai gan vidējais GŪ līmeņa dziļums kūdreņos bija vidēji 55 ± 2 cm, bet purvaiņos vidēji 35 ± 3 cm (starpība vidēji 18 ± 2 cm), *Rkop* mērījumu rezultātos tas neatspoguļojas, jo GŪ līmeņa un *Rkop* mērījumu rezultātiem ir vāja korelācija ($r = 0,3$). Būtiski ($p = 0,002$) atšķiras *Rkop* mērījumu vidējās vērtības mežaudzēs (6,84 ± 0,56 t CO_2-C ha^{-1} gadā) un izcirtumos (10,08 ± 1,96 t CO_2-C ha^{-1} gadā). Visdrīzāk lielākas *Rkop* emisijas izcirtumos veicina mašinizētas mežizstrādes radītie augsnes bojājumi (James & Harrison, 2016) un ciršanas atlieku sadalīšanās (Jandl et al., 2007). Līdzīgas tendences novērotas apmežotā augstajā purvā Skotijā, kur hidromeliorētās platībās un platībās ar saglabātu hidroloģisko režīmu novērtēta gada kopējā *Rkop* ir attie-



3.7. att. **Augsnes kopējās elpošanas mērījumu rezultātu izkliede**
Fig. 3.7. ***Variation of soil total respiration emission measurement results***

cīgi 4,53 t CO_2-C ha^{-1} gadā un 3,35 t CO_2-C ha^{-1} gadā, bet platībās bez meža *Rkop* sasniedz 6,95 t CO_2-C ha^{-1} gadā (Yamulki et al., 2013).

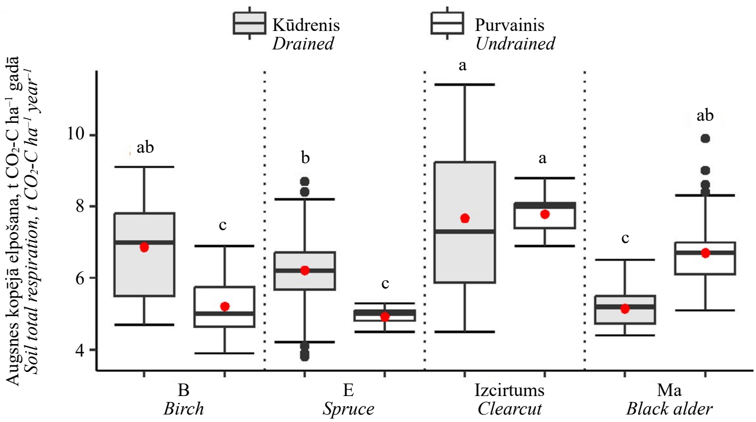
Rkop mērījumu rezultātu variācija galvenokārt skaidrojama ar atmosfēras un attiecīgi arī augsnes temperatūras izmaiņām. Starp *Rkop* un *Ta* mērījumu rezultātiem pastāv cieša korelācija (r = 0,89). Sakarība starp *Rkop* un *Ta* mērījumu rezultātiem raksturojama ar eksponenciālas regresijas vienādojumu (3.8. att.), kas nosaka, ka, *Ta* palielinoties no –1,0 līdz 22,0°C, *Rkop* no 1,5 t CO_2-C ha^{-1} gadā palielinās līdz 29,2 t CO_2-C ha^{-1} gadā. Konstatēts, ka pētījuma laikā GŪ līmeņa dziļumam bijusi vāja (r = 0,30) ietekme uz *Rkop*.



3.8. att. **Sakarība starp augsnes temperatūru un kopējo elpošanu**
Fig. 3.8. ***Relationship between soil temperature and total respiration***

Rkop izcirtumos var būt lielāka nekā mežaudzēs visā pētījumā vērtētajā augsnes temperatūras diapazonā. *Ta* palielinoties no 0 līdz 20°C, *Rkop* CO_2 emisiju izcirtumos un mežaudzēs starpība palielinās no vidēji 1,5 līdz 15,6 t CO_2-C ha^{-1} gadā. Atbilstoši novērotajai *Ta* un CO_2 emisiju mērījumu rezultātu sakarībai (3.8. att.), augsnei esot temperatūrā, kas atbilst LVĢMC noteiktajai gada vidējās gaisa temperatūras klimatiskā standarta normai Latvijā (7°C), tās *Rkop* radītās prognozējamās CO_2 emisijas izcirtumos un mežaudzēs ir attiecīgi 4,9 un 3,7 t CO_2-C ha^{-1} gadā (starpība 1,2 t C ha^{-1} gadā) ar vidējo kvadrātisko kļūdu attiecīgi ± 3,67 un ± 2,4 t CO_2-C ha^{-1} gadā.

Atbilstoši meteoroloģisko staciju Latvijas klimatu raksturojošiem datiem par gaisa temperatūru laika periodā no 2012. līdz 2021. gadam, novērtētā ikgadējā *Rkop* ir no 4,5 līdz 11,4 t CO_2-C ha^{-1} gadā (vidēji 7,70 ± 0,53 t CO_2-C ha^{-1} gadā) izcirtumos, savukārt mežaudzēs no 3,8 līdz 9,9 t CO_2-C ha^{-1} gadā (vidēji 6,14 ± 0,15 t CO_2-C ha^{-1} gadā). Lai gan tika konstatēts, ka *Rkop* mērījumu vidējās vērtības parauglaukumos ar dažādu VKS un MAAT būtiski neatšķiras, ikgadējo emisiju aprēķina rezultāti norāda, ka MAAT un VKS ir būtiska ietekme uz vidējām ikgadējām *Rkop* CO_2 emisijām 10 gadu laikā (3.9. att.). Modelēšanā pielietotie dati norāda, ka ikgadējās gaisa temperatūras mainība individuālos parauglaukumos



3.9. att. **Ikgadējās augsnes kopējās elpošanas variācija**
Fig. 3.9. ***Variation of annual soil total respiration***

gada kumulatīvās *Rkop* emisijas var ietekmēt no 0,3 līdz 3,3 t CO_2-C ha^{-1} gadā (vidēji 1,6 t CO_2-C ha^{-1} gadā). Kas norāda, ka, prognozējot augsnes emisijas valsts mērogā un ilgtermiņā, ir svarīgi ņemt vērā vēsturisko un prognozējamo gaisa temperatūras dinamiku.

Atbilstoši temperatūras variācijai laika periodā kopš 2012. līdz 2021. gadam, individuālos parauglaukumos modelēto ikgadējo *Rkop* CO_2 emisiju variācijas koeficients ir no 1,2 līdz 13,4% (vidēji 8,2%). Tomēr pētījuma gadā un 10 gadu vidējās aprēķinātās ikgadējās *Rkop* emisijas būtiski neatšķiras (3.3. tab.), kas skaidrojams ar to, ka gaisa temperatūras dinamika pētījuma gadā ir bijusi klimata reprezentatīva. Ar šādu pieeju novērtētajām ikgadējām *Rkop* CO_2 emisijām ir salīdzinoši maza nenoteiktība, salīdzinājumam, Somijas Nacionālā SEG inventarizācijā tiek pielietots CO_2 EF ar nenoteiktību 150% (Statistics Finland, 2014).

3.3. tabula / *Table 3.3*

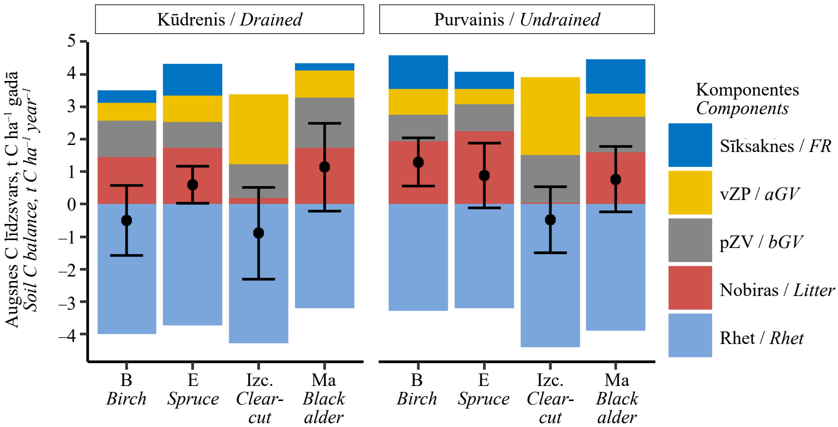
Ikgadējo augsnes elpošanas emisiju novērtējuma kopsavilkums
Summary of annual soil respiration emission assessment

MAAT	VKS [2]	Kopējā elpošana, t CO_2-C ha^{-1} gadā *Total respiration, t CO_2-C ha^{-1} year^{-1}*		Piemērotais *Rhet* īpatsvars, % *Share of Rhet applied, %*	Heterotrofā elpošana, t CO_2-C ha^{-1} gadā *Heterotrophic respiration, t CO_2-C ha^{-1} year^{-1}*	
		Pētījuma gads *Study year*	Klimata dati [1] *Climate data* [1]		Pētījuma gads *Study year*	Klimata dati [1] *Climate data* [1]
Kūdrenis *Drained*	B	6.92 ± 1.58	6.87 ± 0.49	58 ± 4	4.00 ± 0.68	3.97 ± 0.21
	E	6.27 ± 0.52	6.22 ± 0.19	60 ± 1	3.72 ± 0.23	3.7 ± 0.08
	Ma	5.1 ± 0.16	5.16 ± 0.27	63 ± 1	3.21 ± 0.07	3.23 ± 0.12
	Izc.	7.63 ± 2.19	7.68 ± 0.66	57 ± 4	4.28 ± 0.9	4.3 ± 0.27
Purvainis *Undrained*	B	5.27 ± 1.05	5.21 ± 0.31	63 ± 3	3.28 ± 0.47	3.25 ± 0.14
	E	5.1	4.94 ± 0.16	63	3.2	3.14 ± 0.07
	Ma	6.64 ± 0.69	6.71 ± 0.31	59 ± 2	3.89 ± 0.29	3.91 ± 0.13
	Izc.	7.9	7.8 ± 0.4	56	4.4	4.37 ± 0.16

[1] gaisa temperatūra laika periodā no 2012. līdz 2021. gadam / *air temperature during period between 2012 till 2021.*
[2] B – *birch*, E – *spruce*, Ma – *black alder*, Izc. – *clearcut.*

Pētījuma rezultātā aprēķinātā kūdreņu augsnes vidējā ikgadējā *Rhet* 3,80 ± 0,44 t CO_2-C ha^{-1} gadā (no 2,9 līdz 4,4 t CO_2-C ha^{-1} gadā pētījuma mežaudzēs) iekļaujas *Rhet* diapazonā, kas novērtēts citos pētījumos reģionā. *Rhet* ar tiešām mērījumu metodēm visplašāk pētīta Somijas mežos ar sekojošiem *Rhet* rezultātiem: 1,85 ± 0,09 līdz 4,26 ± 0,26 t CO_2-C ha^{-1} gadā no hidromeliorētas organiskās augsnes ar dažādu auglību (Minkkinen et al., 2007); 1,46 līdz 6,70 t CO_2-C ha^{-1} gadā no hidromeliorētas kūdraugsnes (Ojanen et al., 2010); 2,07 līdz 5,39 t CO_2-C ha^{-1} gadā no apmežotas aramzemes organiskas augsnes un 2,76 līdz 4,79 t CO_2-C ha^{-1} gadā no kūdraugsnes rekultivētā kūdras izstrādes laukā (Mäkiranta et al., 2007). Citā pētījumā, kas aptver reģionu no Igaunijas līdz Somijai, novērtēts ka hidromeliorēta meža kūdraugsnes *Rhet* ir no 2,48 līdz 5,15 t CO_2-C ha^{-1} gadā (Minkkinen et al., 2007).

<u>Augsnes neto CO_2 emisijas</u>. Atbilstoši pētījumā ievāktajam empīriskajam materiālam, izmēģinājumu periodā augsne bija neto CO_2 avots purvaiņu izcirtumos (0,49 ± 1,02 t CO_2-C ha^{-1} gadā) un kūdreņu izcirtumos (0,89 ± 0,99 t CO_2-C ha^{-1} gadā), kā arī bērzu kūdreņos (0,50 ± 1,08 t CO_2-C ha^{-1} gadā). Lapkoku purvaiņos un kūdreņos augsne nodrošināja neto CO_2 piesaisti vidēji, attiecīgi, 0,32 ± 0,93 t CO_2-C ha^{-1} gadā un 0,94 ± 1,38 t CO_2-C ha^{-1} gadā, bet egļu audzēs neto piesaisti, attiecīgi, 0,88 ± 1,01 t CO_2-C ha^{-1} gadā un 0,60 ± 74 t CO_2-C ha^{-1} gadā (3.10. att.).



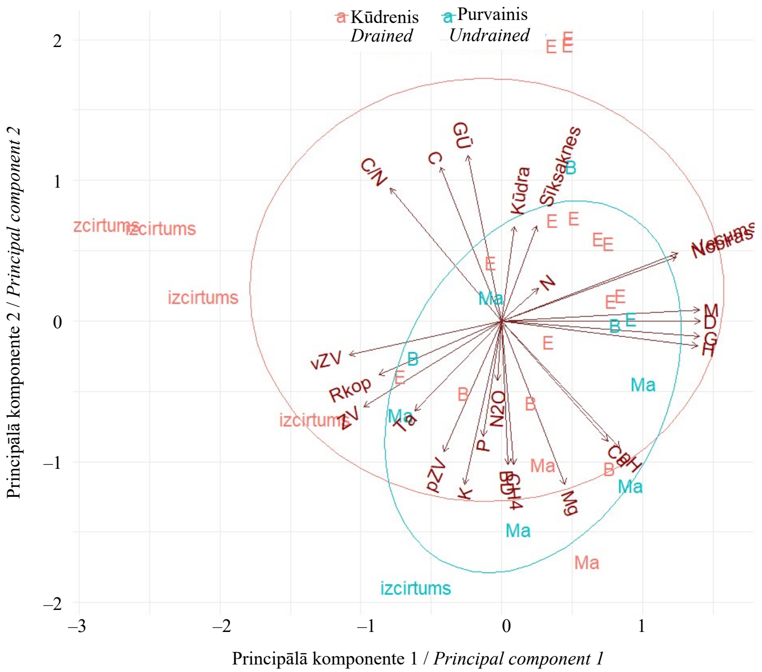
3.10. att. **Augsnes C līdzsvars pētījuma objektos monitoringa gadā**
Fig. 3.10. ***Soil C balance in the monitoring year***

Mežaudzēs ZV un nobirām ir vienlīdz nozīmīga loma augsnes ikgadējās C oglekļa ieneses nodrošināšanā. ZV un vainaga nobiras mežaudzēs ar dažādu VKS pētījuma gadā nodrošināja vidēji attiecīgi 1,72 ± 0,33 t C ha^{-1} gadā un 1,79 ± 0,25 t C ha^{-1} gadā jeb attiecīgi 41 ± 8% un 43 ± 6% no pētījumā novērtētās kopējās augsnes C ieneses. Sīksaknes mežaudzēs nodrošināja C ienesi vidēji 0,71 ± 0,34 t C ha^{-1} gadā jeb 16 ± 7% no kopējās novērtētās augsnes C ieneses. Izcirtumos augsnes C ieneses ar sīksaknēm un nobirām iztrūkums tiecās tikt kompensēts ar lielāku zemsedzes veģetācijas biomasu. Mežaudzēs kopējā novērtētā augsnes C ienese ar ZV bija vidēji 3,47 ± 0,54 t C ha^{-1} gadā, bet izcirtumos ZV nodrošināja augsnes C ienesi vidēji 6,92 ± 0,96 t C ha^{-1} gadā. Attiecīgi, izcirtumos novērtētā ZV biomasa bija aptuveni 2 reizes lielāka nekā mežaudzēs.

Pētījuma novērtējums – augsne mežaudzēs ir neto C piesaistītāja – ir saskaņā ar iepriekšējā Latvijā veiktā pētījuma rezultātiem par augsnes C uzkrājuma izmaiņām kūdreņos (Lupikis & Lazdins, 2017). Tas skaidrojams ar biomasas atmiruma radītu augsnes C ienesi, kas pilnībā spēj kompensēt ikgadējos augsnes elpošanas radītos C zudumus. Pētījumā augsne izcirtumos novērtēta kā CO_2 emisiju avots, jo, salīdzinot ar mežaudzēm, tajās augsnes elpošanas radītās CO_2 emisijas bija lielākas, bet ikgadējā C ienese – mazākā. Lai gan C ienese ar zemsedzes veģetāciju izcirtumos (vidēji 3,55 ± 0,37 t C ha^{-1} gadā) bija ievērojami lielāka nekā mežos (1,65 ± 0,37 t C ha^{-1} gadā), tā nepēja pilnībā kompensēt par vidēji 0,8 t CO_2-C ha^{-1} gadā lielākas augsnes CO_2 emisijas un C ieneses ar vainaga nobirām (mežaudzēs vidēji 1,8 ± 0,5 t C ha^{-1} gadā), un koku sīksaknēm trūkumu (mežaudzēs vidēji 0,71 ± 0,37 t C ha^{-1} gadā).

3.3. SEG emisiju ietekmējošie faktori

Atbilstoši pētījuma mērījumu parauglaukumos vidējo vērtību principālo komponentu analīzes (PCA) rezultātiem (3.11. att.), zemāks vidējais GŪ līmenis sasitās ar lielāku C koncentrāciju un C/N attiecību augsnes virskārtā. To apstiprina arī korelācijas (r = 0,5; p < 0,05) analīze (3.12. att.). Sakarība starp C/N attiecību un GŪ līmeņa dziļumu norāda, ka ilgstoši zema GŪ līmeņa ietekmē kūdras mineralizācijas pakāpe ir augstāka, bet mineralizācijas aktivitāte zemāka. Tas atspoguļojas augsnes

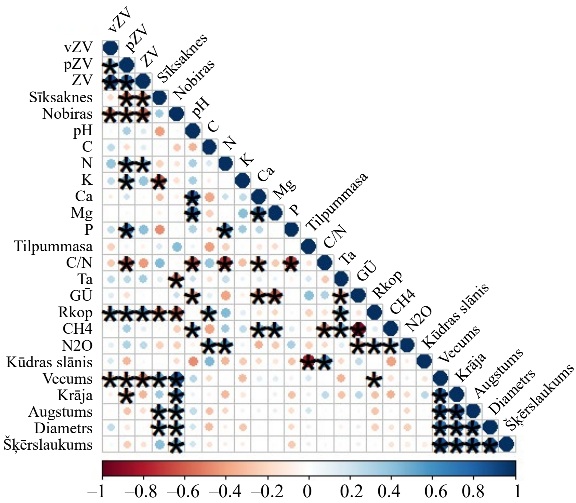


3.11. att. **Mērījumu rezultātu vidējo vērtību principālo komponenšu analīze**

Fig. 3.11. ***Principal component analysis of mean measurement results***

M, D, G, H, vecums – mežaudzes raksturlielumi, attiecīgi – krāja, caurmērs, šķērslaukums, augstums, vecums; pH, Ca, Mg, K, P, C, N, C/N, BD – augsnes raksturlielumi, attiecīgi – vides reakcija, kalcija, magnija, kālija, fosfora, oglekļa un slāpekļa saturs, C un N attiecība; kūdra – kūdras slāņa biezums; sīksaknes, ZV, vZV, pZV, nobiras – ikgadējā C ienese augsnē ar, attiecīgi – sīksaknēm, zemsedzes veģetāciju kopā, virszemes zemsedzes veģetāciju, zemsedzes veģetācijas saknēm, nobirām; Rkop, CH4, N2O – gada kumulatīvās augsnes CO_2, CH_4 un N_2O emisijas; GŪ – gada vidējais attālums no augsnes virskārtas līdz gruntsūdens līmenim; Ta – gada vidējā augsnes temperatūra.

M, D, G, H, age – forest stand characteristics – stock, diameter, basal area, height, age, respectively; pH, Ca, Mg, K, P, C, N, C/N, BD – soil characteristic – acidity, calcium, magnesium, phosphorus, potassium, carbon and nitrogen content, C and N ratio, respectively; peat – thickness of the peat layer; fine roots, ZV, vZV, pZV, litter – annual C input in soil by – FR, GV, aGV, bGV, litter, respectively; Rkop, CH4, N2O – annual cumulative soil CO_2, CH_4 and N_2O emissions; GŪ – annual average GW level; Ta – annual average soil temperature.



3.12. att. **Mērījumu rezultātu vidējo vērtību Spīrmena korelācija**

Fig. 3.12. ***Spearman correlation analysis of mean measurement results***

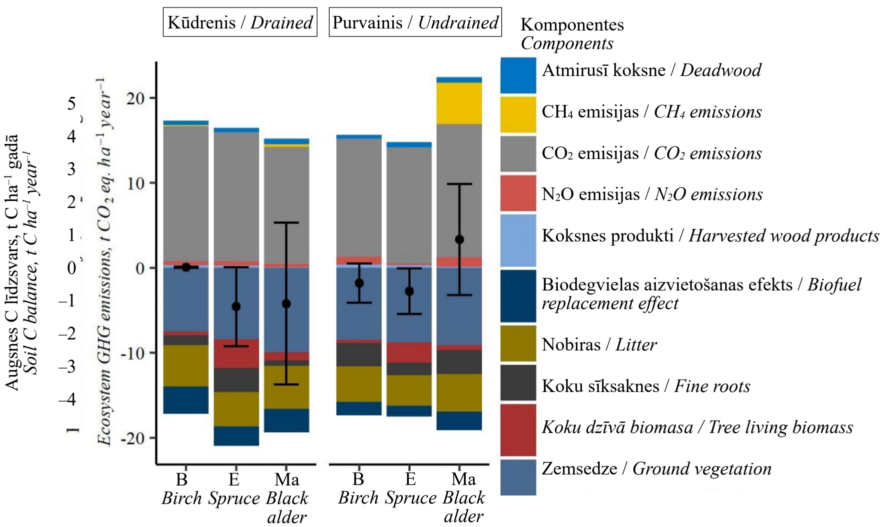
Atšifrējumi atbilstoši 3.11. att. skaidrojumiem; Tilpummasa – *bulk density*; Kūdras slānis – *peat layer depth*; Vecums, Krāja, Augstums, Diametrs, Šķērslaukums – *stand parameters – age, stock, height, diameter, basal area, respectively. See Fig. 3.11 for explanations of other abbreviations.*

SEG emisiju negatīvā korelācijā ar GŪ līmeņa dziļumu. Būtiska negatīva korelācija ir ar CH_4 (ρ = –0,9; $p < 0{,}05$) un N_2O (ρ = –0,4; $p < 0{,}05$), zemāka GŪ līmeņa ietekme visvairāk ietekmē CH_4 emisiju samazināšanos un piesaistes palielināšanos.

Atbilstoši PCA, zemāks vidējais GŪ līmenis, kā arī biezāks kūdras slānis, norāda arī uz lielāku ikgadējo koku sīksakņu produkciju un attiecīgi arī to atmirumu. Kā arī sīksakņu produkcijai ir negatīva sakarība ar pZV biomasu un augsnes auglības rādītājiem – K, Ca, Mg un P koncentrāciju. Korelācijas analīze apstiprina, ka sīksakņu atmirumam ir būtiska negatīva korelācija ar pZV (ρ = –0,6; $p < 0{,}05$) un K koncentrāciju augsnē (Spīrmena un Pīrsona korelācijas koeficients –0,6; $p < 0{,}05$). Kas ir saskaņā ar novērojumu līdzšinējā pētījumā (Lehtonen et al., 2016), ka barības vielu pieejamības trūkumu koki tiecas kompensēt ar lielāku sīksakņu biomasu. Tajā pašā laikā PCA norāda, ka minētajiem augsnes auglības rādītājiem ir tieša sakarība ar augsnes emisijām, kas visizteiktāk ietekmē CH_4 emisijas, bet vismazāk – augsnes elpošanu jeb CO_2 emisijas. Augsnes auglības un tās SEG emisiju sakarības apstiprina arī Spīrmena korelācijas analīze. Proti, CH_4 emisijām ir būtiska korelācija ar augsnes Ca ($r = 0{,}5$; $p < 0{,}05$) un Mg ($r = 0{,}6$; $p < 0{,}05$) koncentrāciju, savukārt N_2O emisijām ir būtiska korelācija ar C ($r = 0{,}5$; $p < 0{,}05$) un N ($r = 0{,}6$; $p < 0{,}05$) koncentrāciju, bet *Rkop* būtiska korelācija konstatēt ar augsnes C koncentrāciju ($r = 0{,}5$; $p < 0{,}05$). Zīmīgi, ka gan uz barības vielu pieejamību, gan augstākām augsnes CH_4 emisijām lielākas augsnes pH vērtības ietekmē norāda gan PCA, gan korelācijas analīze. Zināms, ka augsnes skābums ietekmē metanogēnu un metanofilu populāciju (Serrano-Silva et al., 2014). Turklāt lielāku makroelementu K, Ca, Mg, kā arī P pieejamība atspoguļojas arī lielākā pZV biomasā (3.11. att.). pZV biomasai ir būtiska korelācija ar N un K (r = 0,5; p < 0,05), kā arī P (r = 0,7; p < 0,05) koncentrāciju un C/N attiecību (r = –0,6; p < 0,05) augsnē. PCA norāda arī uz to, ka no vērtētajiem mežaudzes raksturlielumiem, visciešākā sakarība ar ikgadējo vainaga nobiru biomasu ir mežaudzes vecumam. To apstiprina arī korelācijas analīze, lielākais korelācijas koeficients (r = 0,8; p < 0,05) konstatēts sakarībai ar mežaudzes vecumu. Atbilstoši korelācijas analīzei, mežaudzes raksturlielumi nekorelē ar SEG emisijām, bet PCA norāda, ka mežaudzes vecumam ir apgriezta sakarība ar *Rkop*. Proti, augsnes elpošana tiecas samazināties, turpinoties mežaudzes attīstībai, pieaugot tās vecumam (3.11. att.). Tas daļēji skaidrojams mežaudzes vecuma būtisku negatīvu Pīrsona korelāciju ar pZV (r = –0,4; p < 0,05) un vZV (r = –0,6; p < 0,05). Gan PCA, gan Pīrsona korelācijas analīze (r = 0,7; p < 0,05) norāda, ka no zemsedzes veģetācijas komponentēm lielākā ietekme uz autotrofo elpošanu ir tās virszemes biomasai.

3.4. Meža ekosistēmas SEG emisijas

Purvaiņu ar VKS B un E novērtētā ikgadējā vidējā SEG piesaiste ir attiecīgi 1,8 ± 7,57 t CO_2 ekv. ha^{-1} gadā un 2,8 ± 8,3 t CO_2 ekv. ha^{-1} gadā, savukārt novērtētās melnalkšņu purvaiņu ikgadējās vidējās SEG emisijas ir 3,3 ± 13,6 t CO_2 ekv. ha^{-1} gadā. Ma meži novērtēti kā neto SEG emisiju avots galvenokārt pētījumā iegūto augsnes CH_4 emisiju empīrisko datu ietekmē (3.1. tabula). Empīriskie dati norāda, ka melnalkšņu purvaiņu ikgadējās CH_4 emisijas ir vidēji 4,4 ± 3,1 t CO_2 ekv. ha^{-1} gadā, kamēr citās pētījuma objektu grupās ikgadējās novērtētās CH_4 emisijas ir salīdzinoši nenozīmīgas (3.13. att.). Lai gan ekstrēmas augsnes CH_4 emisijas tika konstatētas

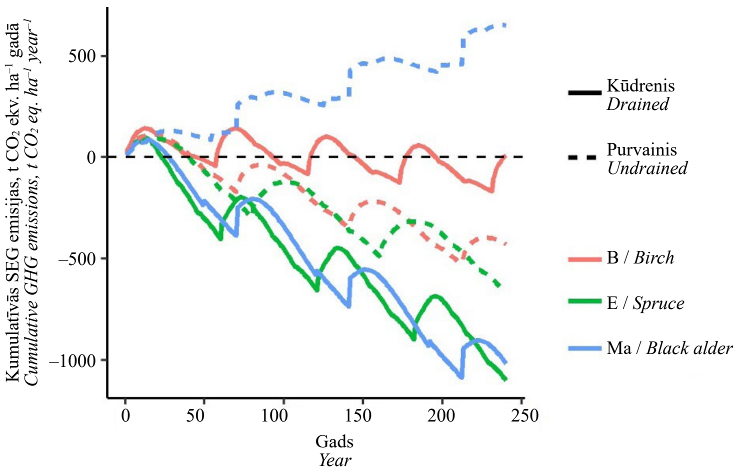


3.13. att. **Meža ekosistēmas ikgadējās vidējās SEG emisijas**

Fig. 3.13. ***Mean annual forest ecosystem GHG emissions***

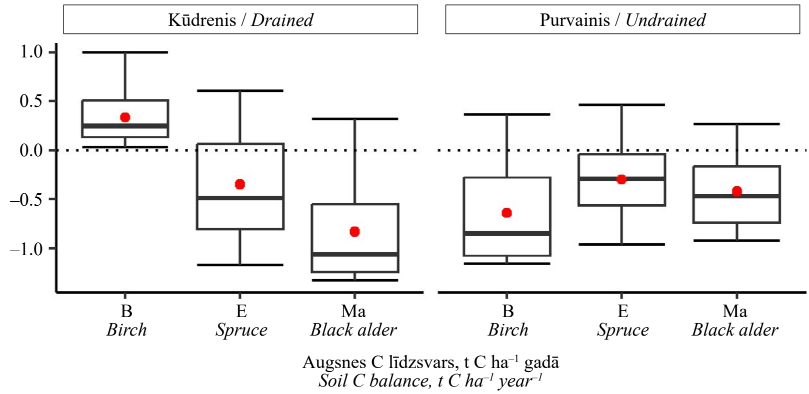
vienā no pieciem parauglaukumiem, kas ierīkoti melnalkšņu purvaiņos, šādu emisiju sastopamības varbūtība purvaiņos nav ignorējama. Kūdreņu ar VKS E un Ma novērtētā ikgadējā vidējā SEG piesaiste ir, attiecīgi, 4,6 ± 12,8 t CO_2 ekv. ha^{-1} gadā un 4,2 ± 17,7 t CO_2 ekv. ha^{-1} gadā. Novērtēts, ka bērzu kūdreņi tiecas būt klimatneitrāli, to vidējās ikgadējās SEG emisijas ir 0,0 ± 11,5 t CO_2 ekv. ha^{-1} gadā.

Ikgadējo SEG emisiju dinamikas kumulatīvā ietekme uz meža ekosistēmas SEG emisijām ilgtermiņā redzama 3.14. attēlā. Pētījumā iegūtie rezultāti norāda, ka meža ekosistēma visu pētījuma objektu grupu sadalījumā (izņemot melnalkšņu purvaiņus) ilgtermiņā vidēji ir neto SEG emisiju piesaistītāja. Tomēr interpretējot novērtētās vidējās ikgadējās vai ilgtermiņa kumulatīvās ekosistēmas SEG emisijas vai to piesaisti, jāņem vērā nenoteiktība. Īpaši piesardzīgi interpretējams kūdreņu ar VKS Ma un B ilgtermiņa kumulatīvo emisiju atspoguļojums 3.14. attēlā. Ņemot vērā empīrisko datu nenoteiktību, ilgtermiņā Ma mežs ikgadēji vidēji var būt gan SEG emisiju avots, gan piesaistītājs (3.13. att.), bet bērzu kūdreņu klimatneitralitātes rezultāts iegūts, aprēķinā izmantojot empīriskos datus ar apvienoto nenoteiktību 134%. Purvaiņu ar VKS E un B kumulatīvo SEG emisiju tendences norāda, ka arī bērzu kūdreņu klimatneitralitātes novērtējums var būt neto SEG emisiju aprēķina komponenšu nenoteiktības kumulatīvā ietekme. Arī aprēķina pieņēmumiem par mežaudžu augšanas gaitu un saimnieciskās darbības intensitāti var būt nozīmīga ietekme uz iegūto meža ekosistēmas neto SEG emisiju aprēķina rezultātu.



3.14. att. **Meža ekosistēmas kumulatīvās SEG emisijas ilgtermiņā**
Fig. 3.14. ***Long term cumulative forest ecosystem GHG emissions***

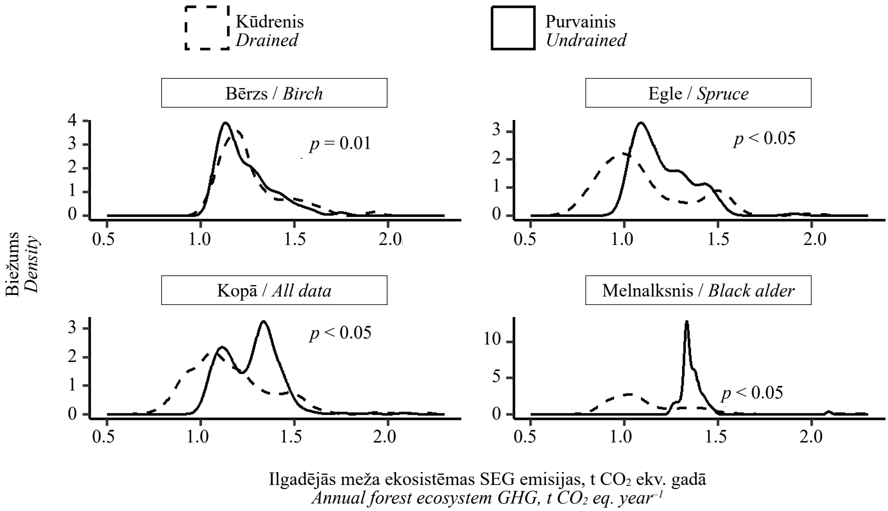
Ņemot vērā ikgadējo augsnes CO_2 emisiju un augsnes C dinamiku meža apsaimniekošanas laikā ilgtermiņā, kas aprēķināta atbilstoši pētījumā iegūtajiem empīriskajiem datiem, kūdreņu un purvaiņu augsne ikgadēji piesaista vidēji attiecīgi 0,28 ± 0,66 t C ha^{-1} gadā un 0,42 ± 0,43 t C ha^{-1} gadā. Purvaiņos ar VKS B, E un Ma augsne ikgadēji piesaista vidēji, attiecīgi, 0,64 ± 0,51 t C ha^{-1} gadā; 0,30 ± 0,33 t C ha^{-1} gadā un 0,33 ± 0,33 t C ha^{-1} gadā, bet kūdreņu augsne, attiecīgi, –0,34 ± 0,26 t C ha^{-1} gadā; 0,35 ± 0,54 t C ha^{-1} gadā un 0,86 ± 0,53 t C ha^{-1} gadā (3.15. att.). Meža apsaimniekošanas cikla laikā ikgadēji augsnes C uzkrājums var gan palielināties, gan samazināties. Pētījumā ievāktie dati norāda, ka to nosaka mežaudzes attīstības stadija. Periodā ar izcirtumu, meža augsne zaudē C, bet mežaudzei attīstoties tā kļūst par C piesaistītāju. Augsnes C uzkrājuma palielināšanos galvenokārt nosaka augsnes C ienese ar vainaga nobirām un sīksaknēm, kas tiecas palielināties līdz ar pieaugošu mežaudzes vecumu.



3.15. att. **Ikgadējās augsnes C piesaistes variācija ilgtermiņā**
Fig. 3.15. ***Variation of annual soil C removals in long term***
Negatīvas vērtības atspoguļo C piesaisti / *Negative values coresponds to C removals.*

Lai gan pētījumā Somijā ir novērtēts, ka ikgadējās augsnes CO_2 emisijas palielinās līdz ar augsnes auglību no 3,8 līdz 12,10 t C ha^{-1} gadā (Ojanen et al., 2010), kas var noteikt, ka augsne ir neto CO_2 avots, promocijas darba pētījuma rezultāti norāda, ka auglīgas organiskas meža augsnes var būt neto CO_2 piesaistītājas. Tas ir saskaņā ar virkni līdzšinējo pētījumu rezultātu, kuri norāda, ka boreālos mežos pēc auglīgas organiskās augsnes hidromeliorācijas to C uzkrājums var ne tikai nemainīties, bet arī pieaugt (Meyer et al., 2013; Varik et al., 2015). Būtisks aspekts, kas var ietekmēt dažādu pētījumu secinājumus ir augsnes C uzkrājuma dinamikas novērtēšana pielietotā metodika, kas var ņemt vai neņemt vērā dažādas augsnes C ieneses komponentes (Ojanen et al., 2012).

Atbilstoši pētījumā ievāktajiem empīriskajiem datiem un meža ekosistēmas SEG emisiju aprēķinu metodikai 240 gadu meža zemes apsaimniekošanas cikla laikā purvaiņi piesaista vidēji 0,2 ± 9,7 t CO_2 ekv. ha^{-1} gadā, bet kūdreņi piesaista vidēji 2,9 ± 14,4 t CO_2 ekv. ha^{-1} gadā. Kūdreņu un purvaiņu ikgadējo SEG emisiju vērtību izkliede ir būtiski atšķirīga (3.16. att.). Tādējādi rezultāti norāda, ka kūdreņi var nodrošināt lielāku ieguldījumu klimata izmaiņu mazināšanā.



3.16. att. **Ikgadējo meža ekosistēmas SEG emisiju vērtību izkliede**

Fig. 3.16. ***Density of annual forest ecosystem annual GHG emissions***

Logaritmiski traksformēti dati / *Logarithmically transformed data.*

SECINĀJUMI

1. Egļu audzēs konstatēta lineāra sakarība starp šķērslaukumu un ikgadējo C ienesi augsnē ($r = 0,9$). Bērzu un melnalkšņu audzēs, šķērslaukumam sasniedzot aptuveni 10 m^2 ha^{-1}, ikgadējā C ienese augsnē strauji palielinās līdz apjomam, kāds egļu audzēs prognozējams ar šķērslaukumu 30 m^2 ha^{-1}. Tādējādi lapu koku audzes ar mazāku šķērslaukumu var nodrošināt lielāku C ienesi augsnē nekā egļu audzes un apsaimniekotos mežos potenciāli sniegt lielāku ieguldījumu augsnes C uzkrājuma saglabāšanā.
2. Zemsedzei ir nozīmīga loma augsnes C uzkrājuma saglabāšanā izcirtumos, jo zemsedzes ikgadējā C ienese augsnē potenciāli var kompensēt C ieneses trūkumu ar vainaga nobirām un koku sīksaknēm. Novērtētā C ienese augsnē ar zemsedzi analizētajos izcirtumos bija būtiski lielākā nekā mežaudzēs ($p < 0,05$), attiecīgi 3,3 ± 0,5 un 1,7 ± 0,3 t C ha^{-1} gadā. Arī zemsedzes biomasas un audzes vecuma sakarība ($r = –0,6$) norāda, ka zemsedzes biomasa izcirtumos ir vidēji divas reizes lielāka nekā 80 gadus vecās audzēs.
3. Nav konstatēta būtiska audžu vecuma, caurmēra un krājas ietekme uz ikgadējo koku sīksakņu atmirumu (vidēji 1,5 ± 0,8 t ha^{-1} gadā).
4. Analizējot sakarību starp gada kopējām augsnes CH_4 emisijām un vidējo gruntsūdens līmeni ($r = –0,6$) noskaidrots, ka augsne ir CH_4 emisiju avots, ja vidējais gruntsūdens līmenis ir augstāk par 30 cm. Lai gan gruntsūdens līmeņa un augsnes CH_4 emisiju mērījumu sakarības purvaiņos un kūdreņos ir līdzīgas, ievērojami paaugstinātu emisiju sastopamības varbūtība purvaiņos nosaka, ka emisiju prognozēšanā ir svarīgi novērtēt arī faktisko hidromeliorācijas sistēmas funkcionalitāti.
5. Konstatēta vidēji cieša sakarība ($r = –0,4$) starp gruntsūdens līmeņa mērījumu vidējām vērtībām un gada kopējām augsnes N_2O emisijām. Gada kopējās augsnes N_2O emisijas kūdreņos (vidēji 1,1 ± 0,4 kg N ha^{-1} gadā) un purvaiņos (vidēji 2,6 ± 0,9 kg N ha^{-1} gadā) ir būtiski atšķirīgas ($p < 0,01$).
6. Novērtētās, Latvijas klimatiskajiem apstākļiem raksturīgās ikgadējās augsnes kopējās elpošanas CO_2 emisijas analizētajos izcirtumos (vidēji 7,7 ± 0,5 t C ha^{-1} gadā) ir lielākas ($p < 0,05$) nekā mežaudzēs (vidēji 6,1 ± 0,2 t C ha^{-1} gadā). Nav konstatēta būtiska hidromeliorācijas vai valdošās koku sugas ietekme uz augsnes kopējās elpošanas CO_2 emisijām.
7. Oglekļa uzkrājuma zudumi eitrofu purvaiņu un kūdreņu izcirtumu augsnē (vidēji 0,7 t C ha^{-1} gadā) meža apsaimniekošanas cikla laikā tiek kompensēti ar C piesaisti audzēs (vidēji 0,6 t C ha^{-1} gadā).
   Meži ar hidromeliorētu augsni var nodrošināt lielāku ieguldījumu klimata pārmaiņu mazināšanā, jo intensīvi apsaimniekotu eitrofu kūdreņu un purvaiņu ekosistēma piesaista, attiecīgi, vidēji 2,9 un 0,2 t CO_2 ekv. gadā.

1. GENERAL DESCRIPTION OF THE THESIS

1.1. Topicality

The topicality of this study is determined by the Paris Agreement and related international regulatory acts, which stipulate that after 2050, the land use, land use change, and forestry (LULUCF) sector must compensate for Latvia's total greenhouse gas (GHG) emissions. Organic forest soils, particularly peat and peaty soils in Latvia, are a significant source of GHG emissions in the country, and one of the most effective climate change mitigation measures in the LULUCF sector is related to their management. However, there is currently a lack of knowledge on the potential contribution of forests with different nutrient availability organic soil management scenarios to mitigating climate change. In the national GHG inventory, a single carbon dioxide (CO_2) emission factor (EF) obtained from national studies is applied to calculate the CO_2 emissions from drained organic soil, regardless of its nutrient status. For the calculation of methane (CH_4) and nitrous oxide (N_2O) emissions, unverified EFs developed in studies in a temperate climate zone are used in the national GHG inventory. This study aims to develop GHG EFs for drained and undrained nutrient-rich organic forest soils and to estimate the net GHG emissions of the forest ecosystem with such soils. The acquired knowledge can be used to improve the national GHG inventory methodology and to plan climate change mitigation measures.

1.2. Research aim, objectives and thesis

The aim of this thesis is to assess the total greenhouse gas (CO_2, CH_4, N_2O) emissions of the soil and forest ecosystem with both drained and undrained nutrient-rich organic soil. The following research objectives have been established:

1. to develop coefficients that characterize the soil carbon input from foliar litter, fine roots of trees, and ground vegetation in spruce (*Picea abies* (L.) H. Karst), birch (*Betula* spp.), and black alder (*Alnus glutinosa* (L.) Gaertn) forests with both drained and undrained nutrient-rich organic soil.
2. To develop emission factors for the estimation of CO_2, CH_4, and N_2O emissions from drained and undrained nutrien-rich organic soil in spruce, birch and black alder forests.
3. To estimate the net GHG emissions from spruce, birch, and black alder forests with both drained and undrained nutrient-rich organic soil.

The research thesis:

1. The carbon stock of drained and undrained nutrient rich organic forest soil is not decreasing.
2. Forest ecosystems with drained and undrained nutrient-rich organic soil are not a net source of GHG emissions.

1.3. Scientific novelty and practical significance of the work, recommendations

Previous studies on soil GHG emissions in managed forests have primarily focused on drained areas, and the study results provide a limited geographical representativeness. Most of the previous studies have been conducted in forests of Finland with drained organic soils of varying nutrient availability. As a result, research on GHG emissions in hemiboreal forests is insufficient. Previous studies have primarily focused on evaluating changes in soil carbon (C) stocks or GHG emissions only during the monitoring period of the study. Lack of understanding of the dynamics of soil C stock and ecosystem GHG emissions during the forest management cycle still persists. Studies on undrained organic soil are rare due to insufficient motivation arising from the methodology of GHG inventory, which only requires reporting GHG emissions from drained organic soils. The GHG inventory approach does not provide a comprehensive understanding of the impact of forest organic soil drainage or preservation of the natural soil moisture regime on a country's total GHG emissions. Therefore, quantifying emissions produced by undrained soils is as crucial as assessing emissions produced by drained soils. The results of the research presented in this doctoral thesis contribute to filling these critical knowledge gaps.

The study evaluates the net GHG emissions of undrained and drained nutrient-rich organic soils, and of the associated forest ecosystem, over the course of a forest management cycle. The research aims to gain a better understanding of how forest drainage affects GHG emissions and enable comparison between forest management scenarios with and without soil drainage. The developed factors and equations for GHG emission estimation can contribute to refining the GHG inventory methodology for hemiboreal forests.

<u>Recommendations:</u>

1. It is recommended to use the following research results to estimate the annual soil C input in nutrient-rich organic soil:
   a. regression equations that describe the C input from deciduous or spruce forest foliar litter based on the basal area of the forest stand;
   b. coefficients for the C input by ground vegetation (GV) biomass in clearcuts and stands dominated by birch, black alder, and spruce, taking into account the soil drainage status;
   c. and coefficients for the annual mortality of fine root biomass of trees (FR, fine roots of trees) in deciduous and spruce forests, based on the soil drainage status.

   To enhance the accuracy of soil C input forecasting and evaluate the uncertainty of the results, extensive long-term studies with sampling of biomass of FR and GV in large number of replicates are required on a nationwide level in Latvia.
2. For calculation if N_2O emissions from the soil, EF provided by the study can be used. The calculation of CH_4 emissions should consider the average groundwater (GW) level in the area, as well as the likelihood of extreme emissions. Therefore

for estimation of CH_4 emissions equation characterizing emissions depending from average GW level taking into account occurrence of extreme emissions should be used. A threshold soil temperature of –5°C can be used to account for insignificant emissions during winter. However, care should be taken when calculating CH_4 emissions in areas with shallow mean groundwater level (less than 30 cm), as the uncertainty of the results increases significantly. Further research is needed in such areas to reduce this uncertainty.

3. The variability of intra-annual air temperature can have an average impact of ± 1,6 t C ha^{-1} $year^{-1}$ on the annual cumulative CO_2 emissions by soil respiration, therefore it is important to consider regional and annual air temperature variability in long-term and national-scale predictions of soil emissions. To accurately predict annual cumulative soil respiration emissions, it is recommended to use an interpolation approach based on hourly mean air temperature, as using daily mean air temperature for calculation can potentially overestimate the emissions by an average of 1,5 t C ha^{-1} $year^{-1}$. Similarly, the annual meteorological conditions' impact on the annual mean GW level should be considered when forecasting CH_4 emissions.
4. The annual GHG emissions and CO_2 removals by the forest ecosystem are influenced by the development stage of the forest stand, forest management activities, and meteorological conditions. It is important to consider these factors when evaluating and comparing GHG emissions across different forest management scenarios.

1.4. Dissemination

The research results have been published in seven scientific articles and presented in seven international scientific conferences (Chapter 1.4).

1.5. Structure of the doctoral thesis

The structure of the thesis aligns with the research tasks established in the study. The work is divided into three chapters. The first chapter provides an overview of the current knowledge on GHG emissions and C cycling in forest organic soils. The second chapter outlines the methods used to collect and analyse empirical data. The third chapter presents and discusses the results of the study according to the doctoral thesis's objectives and research tasks.

The volume of the thesis: 106 pages, 19 tables, 39 figures, 5 annexes, and 296 references. The conclusion of the study presents nine key findings and provides four recommendations.

2. MATERIALS AND METHODS

The empirical data was collected in 31 forest compartments between October 2019 and May 2022. Each forest compartment was represented by one sample plot (Figure 2.1), where the characteristics of the stand, depth of the GW level, soil and atmospheric temperature were measured, and soil GHG emission, soil, soil water, foliar litter, GV and FR biomass samples were taken for analysis in the laboratory. Data collection took place over a period of 12 consecutive months at each sample plot.

Samples of foliar litter, GV, and FR biomass were collected and their dry matter and C content were determined to estimate the annual input of C into the soil. The results of soil GHG emission measurements, soil C input, and the factors affecting them were analysed to recognize and describe GHG emissions from forests with nutrient-rich organic soils and to examine the relationships between them.

2.1. Study site desription

To characterize GHG emissions and the affecting factors in forests with both drained and undrained nutrient-rich organic soil, sample plots were established in 21 drained (*Oxalidosa turf. mel.*) and 10 undrained (*Dryopterioso-caricosa, Filipendulosa*) forest sites of varying developmental stages. The study included 10 to 80-year-old forest stands (26 stands) and five clearcuts. All sample plots were located within 30 km of meteorological stations of the Latvian Environment, Geology and Meteorology Centre (LVĢMC) (Figure 2.1). The drained sites consisted of three silver birch (birch, *Betula* spp.), two black alder (*Alnus glutinosa* (L.) Gaertn.), 12 Norway spruce (spruce, *Picea abies* (L.) H. Karst.) stands, and four clearcuts, while the undrained sites consisted of three birch, five black alder, one spruce stand, and one clearcut. Soil and soil water samples were collected and analysed in the laboratory, and GW level measurements were taken to characterize the research objects. Additionally, the characteristics of the tree stands were determined. The results of these measurements and analyses were used to identify the factors affecting soil GHG emissions and to characterize the relationships between them.

C content in the upper 20 cm of soil in the study sites ranges from 342 to 507 g C kg^{-1} (mean 455 ± 43 g C kg^{-1}) in undrained soil and 328 to 569 g C kg^{-1} (mean 487 ± 40 g C kg^{-1}) in drained soil. The mean carbon-to-nitrogen (C/N) ratio, regardless of soil drainage status, was 19 ± 3, with a range of 13 to 31 in individual research sites. The mean values also of other evaluated chemical elements, pH, and soil density did not differ significantly between drained and undrained soil. The mean values of soil chemical parameters were: 0.5 ± 0.1 g K kg^{-1}, 21.0 ± 4.5 g Ca kg^{-1}, 2.1 ± 0.4 g Mg kg^{-1}, and 1.3 ± 0.4 g P kg^{-1}. The average soil density was 426.0 ± 29.3 kg m^{-3}, and the average soil pH was 4.5 ± 0.4.

The average distance of GW level from the soil surface was 55 ± 2 cm in drained sites and 35 ± 3 cm in undrained sites. The monthly average GW level in forests with drained soil was, on average, 18 ± 2 cm lower from the soil surface compared to the GW level in forests with undrained soil (Figure 2.2).

2.2. Collection and analysis of foliar litter, ground vegetation and fine root samples

Tree foliar litter was captured using five cone-shaped litter traps with a surface area of 0.5 m^2, evenly spaced within each plot. The litter was collected for 12 consecutive months with an interval of 4 weeks and brought to the laboratory. The dry matter of each sample was determined – all tree foliar litter components were weighted, including needles, leaves, wood, bark, cones, seeds, and lichens, excluding branches longer than 10 cm. During the entire 12-month period, the litter collected from a single litter trap was pooled and ground into a fine powder for determination of C content.

GV samples were collected in each plot in four replicates, by separately taking samples of above-ground (aGV) and below-ground (bGV) GV biomass. A 1 m^2 square area was selected in each plot with the vegetation characteristic of the forest stand. Sampling was done in four smaller square-shaped plots (side length 20 cm) located at the corners of the selected 1 m^2 square. To obtain the aGV biomass sample, all the living vegetation of herbaceous plants (herbs and grasses) plants within the soil surface was collected. The bGV biomass sample was obtained by collecting the roots of the ground vegetation from the upper 20 cm soil layer. The samples were collected in August, when the maximum amount of GV biomass was expected (Uri et al., 2017). In the laboratory, the bGV root biomass was cleaned of soil particles and tree roots by rinsing (wet sieving) and sorting by root morphological characteristics.

FR (diameter less than 2 mm) production samples were collected using a modified fine root ingrowth core method (Laiho et al., 2014; Bhuiyan et al., 2017). The method involves the use of a flexible polyester cylindrical mesh bag (ingrowth cylinder), 80 cm long, 35 mm in diameter, and with a mesh size of 2 × 2 mm, placed in a 60 cm deep hole in the peat. In each plot, before the beginning of the vegetation season, six ingrowth cores were installed, spaced one meter apart from the centre of the plot to its outer border. Soil for filling the core was obtained near its installation point using a soil auger. Half of the cores were removed after the end of one vegetation season and the remaining half after two vegetation seasons. The ingrowth cores were carefully removed from the soil to avoid pulling out the ingrown roots and were taken to the laboratory. In the laboratory, the ingrown roots were trimmed from the outside along with the surface of the core, and the fine roots within the cylinder were separated from the soil by wet sieving. Before determining the dry matter of the FR, only the roots of trees were sorted by morphological characteristics from all of the sampled roots.

The biomass dry matter content of foliar litter, GV, and FR samples was determined by oven-drying the samples at 70°C until a constant mass was achieved and then weighing them. The C content of the litter and GV samples was then determined by grinding the samples into a fine powder and using the dry combustion method (elemental analysis).

2.3. Collection and analysis of soil greenhous gas emissions samples

The sampling of soil GHG emissions was performed using the manual closed chamber method (Hutchinson & Livingston, 1993). The GHG emission sampling set consisted of a collar and a gas sampling chamber made of polyvinyl chloride (PVC) material. The collar had a diameter of 50 cm, and the chamber had a height of 40 cm, and a volume of 63 litres. In each plot, five permanent soil collars were installed at a depth of 5 cm and at least one month prior to the first GHG emission sampling. During collar installation root disruption was avoided and the GV and litter layer was preserved (Pavelka et al., 2018). The ground vegetation and litter layer were kept intact throughout the GHG emission monitoring process. Therefore, the collected gas samples reflect the total respiration of the soil (*Rtot*), which is comprised of both the heterotrophic respiration of the soil contained within the chamber and the autotrophic respiration of the above-ground vegetation and biomass in the soil.

Soil GHG emissions were sampled for a period of 12 months, with intervals of four weeks between plot surveys. The soil GHG samples were taken by placing the chamber on top of the soil collar, and collecting four gas samples within 30 minutes, at a 10-minute interval. The gas samples were taken with 100 mL glass bottles with underpressure of 0.3 mbar and were transported to the laboratory for analysis. The gas content (CO_2, CH_4, and N_2O) was determined using a Shimadzu Nexis GC-2030 gas chromatograph (Loftfield et al., 1997).

In addition to the soil GHG emissions sampling, the temperature and GW level were also measured in each sample plot. The temperature was measured for both the air and soil at a depth of 5 cm (*Ts*) near the soil collar (Pavelka et al., 2018). The GW level was determined using a measuring tape and previously installed PVC pipes that reached a depth of 140 cm in each sample plot. These measurements were taken simultaneously with the soil GHG emissions sampling to provide additional information about the environmental conditions affecting GHG emissions.

2.4. Estimation of soil carbon input by foliar litter, ground vegetation and fine roots

The annual C input to the soil is calculated by extrapolating the measured C sequestered by the net ecosystem productivity of foliar litter, FR production, and GV to a 1-hectare area (2.1). The calculations are based on several assumptions:

- the GV biomass estimated at the end of the growing season in August is equal to its annual net ecosystem productivity and annual mortality;
- the annual FR production is equal to the difference in the biomass of roots grown into the ingrowth core in two and one growing season (Bhuiyan et al., 2017);

- the production of FR determined by the ingrowth core method is equal to the annual mortality of FR (Laiho et al., 2014);
- all the C from the annual foliar litter, FR production, and GV biomass is annually transferred to the soil C stock.

$$C_{input} = \frac{m_{input} \bullet 10000}{S} \bullet \frac{C}{100} \text{ , where} \tag{2.1}$$

C_{input} – annual soil C input by foliar litter, FR or GV (aGV or bGV), t C ha^{-1} year^{-1};
m_{input} – the biomass of the litter collected from the litter traps during the year, the annual production of FR or the biomass of the GV (aGV or bGV) collected from the sampling site, dry matter t;
S – area of litter trap, cross-sectional area of root ingrowth core or area of GV sampling location, m^2;
C – C concentration in oven-dry foliar litter, FR, aGV or bGV biomass, %.

2.5. Elaboration of soil greenhous gas emission estimation factors and equations

For the calculation of soil GHG emissions, a linear regression analysis was initially performed using data on the concentration of GHG in the chamber immediately after the installation of the chamber on the collar and 10; 20; 30 minutes after taking the first sample. In order to ensure the reliability of the obtained results, logical data control was performed to exclude data that did not follow a linear change in gas concentration in the regression analysis. In addition, the coefficient of determination (R^2) of each acquired linear regression equation was evaluated, and for the subsequent calculation of soil GHG emissions only slope coefficients (variable "b" in Equation 2.2) from equations with $R^2 > 0.7$ were used, except when the estimated difference between maximum and minimum GHG concentrations in the chamber is smaller than the uncertainty of the applied gas chromatographic sample testing method. The obtained slope coefficients of the linear equations, which characterize the change in the GHG concentration in the chamber during gas sampling, were used to calculate the *Rtot*, by the equation of ideal gas law:

$$GHG = \frac{M \bullet P \bullet V \bullet b}{R \bullet T \bullet S} \text{ , where} \tag{2.2}$$

GHG – GHG exchange between the atmosphere and the soil, including the vegetation contained in the chamber, µg GHG m^{-2} h^{-1};
M – molar mass of GHG, g mol^{-1};
P – air pressure in the chamber = 101,300 Pa;
V – chamber volume = 0.063 m^3;
b – the slope coefficient of the linear regression equation, which characterizes the change in gas concentration in the chamber per unit of time, ppm h^{-1};
R – universal gas constant = 8.314 m^3 Pa K^{-1} mol^{-1};
T – air temperature, K;
S – soil collar area = 0.1995 m^2.

It is assumed that CH_4 and N_2O exchange estimated by Equation 2.2 is equal to soil CH_4 and N_2O emissions. In the calculation of annual total soil CH_4 and N_2O emissions, it is assumed that the results of monthly emission measurements are equal to the total soil emissions of the relevant month in the sampling plot. Accordingly, the annual soil GHG emissions are calculated as the sum of the monthly soil GHG emissions:

$$GHG_{annual} = \sum GHG_{monthly}\,(Jan...Dec)\text{ , where} \tag{2.3}$$

GHG_{annual} – annual soil GHG emissions, kg ha^{-1} year^{-1};
$GHG_{monthly}$ – monthly total soil GHG emissions, kg ha^{-1} month^{-1}.

The annual cumulative *Rtot* was calculated by interpolating the results of the monthly soil CO_2 emissions measurements using:

- R_{10} and Q_{10} parameters (Varik et al., 2015; Uri et al., 2017; Kriiska et al., 2019);
- the relationship between air temperature and *Ts*;
- and hourly average air temperature data from the closest LVĢMC meteorological station to the individual sample plot.

R_{10} is the rate of *Rtot* at a soil temperature of 10°C and Q_{10} is the factor that describes the change in *Rtot* with every 10°C increase in soil temperature. The values of R_{10} and Q_{10} are calculated based on empirical data collected from the individual sample plots. At first in the process of calculating the interpolated emissions data The coefficient *b* of the exponential equation (2.4), which describes the relationship between soil temperature and *Rtot*, is determined.

$$Rtot = ae^{bTs}\text{ , where} \tag{2.4}$$

Rtot – soil total respiration, µg CO_2-C m^{-2} s^{-1};
a, *b* – coefficients of an exponential equation;
Ts – soil temperature at a depth of 5 cm, °C.

Then the coefficient *b* of the exponential Equation 2.4 is used to calculate the value of Q_{10} (2.5):

$$Q_{10} = e^{10b}\text{ , where} \tag{2.5}$$

Q_{10} – soil respiration temperature sensitivity coefficient;
b – coefficient of an exponential equation.

The R_{10} value estimated by Equation 2.4 and the Q_{10} value estimated by Equation 2.5, combined with *Ts* data, were used to interpolate *Rtot* (2.6) at each plot. Changes in *Ts* over time were determined using a regression equation that describes the relationship between *Ts* and air temperature measurements in the study plots and data on the hourly average air temperature from LVGMC meteorological stations.

$$Rtot = R_{10}\, Q_{10}^{((Ts-10)/10)}\text{ , where} \tag{2.6}$$

Rtot – soil total respiration, µg CO_2-C m^{-2} s^{-1};
Ts – soil temperature at a depth of 5 cm, °C;

R_{10} – total respiration of the soil at *Ts* of 10 °, µg CO_2-C m^{-2} s^{-1};
Q_{10} – soil respiration temperature sensitivity coefficient.

To recalculate the obtained *Rtot* result to soil heterotrophic respiration (*Rhet*), the equation developed in previous studies (Bond-Lamberty et al., 2004) is applied. The equation is suitable for the recalculation of *Rtot* emissions estimated in individual plots.

$$\ln(Rhet) = 1.22 + 0.73\ln(Rs) \quad R^2 = 0.81 \quad P < 0.001 \text{ , where} \qquad (2.7)$$

Rhet – soil heterotrophic respiration, g C m^{-2} $year^{-1}$;
Rs – total soil respiration without above-ground autotrophic respiration, g C m^{-2} $year^{-1}$.

2.6. Evaluation of ecosystem greenhouse gas emissions

The assessment of GHG emissions of the forest ecosystem was done by modelling the dynamics of GHG emissions and CO_2 removals over a 240-year forest management period. This assessment considered:

- annual soil GHG emissions and soil C input from foliar litter, aGV and bGV biomass and FR mortality;
- results from previous studies on annual soil C input from the mortality of mosses and dwarf shrubs;
- C sequestration in living and dead biomass of wood based on the development of the forest stand including annual growth, natural mortality and logging. The data was provided by national forest inventory and the long-term forest resource forecasting model (AGM) of LSFRI "Silava";
- C sequestration in harvested wood products and the effect of biofuel replacement based on the structure of round wood produced in logging and the proportion of biofuel;
- CH_4 emissions from drainage ditches according to results reported by Vanags-Duka et al. (2022);
- indirect soil CO_2 emissions from DOC leaching according to IPCC default EFs.

The assessment was carried out in accordance with the approach of the GHG inventory and methodology of IPCC guidelines for estimation of GHG emissions in forest land remaining forest land category. Accordingly, the calculation of the dynamics of forest GHG emissions is based on the assessment of the annual C stock changes in various sinks (soil, living tree biomass, dead wood and harvested wood products), as well as soil CH_4 and N_2O emissions from soil and drainage ditches. The assessment of the dynamics of annual C removals and GHG emissions was carried out using AGM data on the dynamics of forest growth and logging of spruce, pine and black alder forests with drained and undrained nutrient-rich organic soil. The annual variability of the age of the forest stand and the annual height, diameter and growing stock of living, dead, as well as felled trees within the forest management cycle were used as input values in the amassment.

Soil C stock changes were estimated by summing the annual C loss as a result of *Rhet* and the annual soil C input by foliar litter, GV, FR, mosses and dwarf shrubs. In the calculation of annual soil C loss by *Rhet* and soil C input with GV and FR, fixed values obtained as a result of the study are applied according to dominant tree species and forest land status (forest stand or clearcut) that is distinguished according to the assumption of the dynamics of the annual basal area of managed forests. The threshold of basal area for land status classification was 6; 4 and 5 m^2 ha^{-1} for spruce, birch and black alder forests, respectively. Thus, the fixed values of the soil C loss and input assessed in the research clearcuts are applied in the calculation, if the basal area of the forest stand is below the threshold, while the values obtained in the study sites with forest stands are applied if basal area is greater than the threshold value.

In the calculation of the annual soil C input with tree foliar litter, mosses and dwarf shrubs, equations are used that describe the C input depending on the assumption about the dynamics of the basal area or age of the forest stand during the forest management cycle. Equations developed in the study, which describe the annual C input depending on the basal area of the forest stand in spruce and deciduous forests, were used in the calculation of C input with tree foliar litter. While, in the calculation of the annual C input by mosses and small shrubs, equations developed in previous studies (Muukkonen & Mäkipää, 2006) were used, which characterize the biomass depending on the age of the forest stand. It is assumed that: annual mortality rates of aboveground biomass of dwarf shrubs and mosses are 25% and 33%, respectively (Muukkonen & Mäkipää, 2006) with an average C content of 47.5% (FAO, 2015); 70% of the total C input by shrubs and mosses comes from the above-ground biomass (Mälkönen, 1974; Havas & Kubin, 1983; Palviainen et al., 2005).

It was assumed that the area covered by the drainage ditches was 3% and the CH_4 emissions amounted to 10.3 kg CH_4 ha^{-1} $year^{-1}$ (Vanags-Duka et al., 2022). The emissions related to DOC in drained and undrained areas were calculated using EFs of 1.1 and 0.9 t CO_2 ha^{-1} $year^{-1}$, respectively, and an uncertainty of 66.7% (Hiraishi et al., 2014).

The calculation of the change in C stock in the biomass of trees was determined based on: allometric equations applicable for individual trees (Liepa et al., 2018); as well as AGM data on the dynamics of live, dead, and felled trees in the forest management cycle; and weighted average C content of tree biomass (Bārdule et al., 2021c). The annual change in the C stock of living biomass was determined by subtracting the C stock of the previous year from the C stock in the year of estimation, excluding the C stock of harvested trees and the C stock of decayed trees. When tree is felled it is assumed that its root and crown C stock is transferred from living biomass to deadwood. While the stem C of harvested trees is used to calculate the C input in the pool of harvested wood products. In cases of natural mortality, all tree biomass associated C stock transfers from living biomass to deadwood pool. The C stored in deadwood is assumed to return to the atmosphere within

20 years. To avoid bias in the calculation, it was assumed that the C stock in deadwood at the beginning of the 240-year forest management cycle corresponds to the average C stock in deadwood in the forest, based on the MRM data. The annual change in the C stock of deadwood was calculated as the difference between the C stock in deadwood in the calculation year and the previous year, considering 5% of the C stock that returns to the atmosphere in the calculation year.

The calculation of the annual changes in the C stock of harvested wood products considers the structure of roundwood types produced through harvesting. A 50% loss in C stock due to the production process is assumed for sawn timber and pulpwood. The calculation is performed as the difference between the C stock at the end and beginning of the calculation year, or at the end of the previous year, following the methodology outlined in the National GHG Inventory (Skrebele et al., 2021). The C stock in harvested wood products at the start of the 240-year forest management cycle is assumed to be equal to the average C stock reported in the GHG inventory (Skrebele et al., 2021).

The calculation of the biofuel replacement effect is based on the assumption that share of wood corresponding to annually C leaving the C stock of the category of harvested wood products, the C of wood product production losses, as well as the share of harvested tree trunks in the category biofuel is used as a fuel that replaces the amount of unburned natural gas equivalent in energy value. Thus, according to the assumptions about the energy value of the fuel and the amount of GHG emissions attributed to it depending on the type of fuel (Eggleston et al., 2006), the effect of replacing natural gas with biofuel on GHGs that do not enter the atmosphere is estimated and result included in the forest ecosystem GHG balance.

2.7. Data processing

The relationships between soil C input, GHG emission measurement results and affecting factors were assessed by regression analysis, while the strength of the relationship was evaluated by correlation analysis, determining the Pearson (r) and Spearman (ρ) correlation coefficients. The coefficient of determination (R^2) and the root mean square error (RMSE) were used to characterize the quality of the regression equations. The balance of GHG emissions and CO_2 removals of the ecosystem or individual C pools is expressed in CO_2 equivalents, CH_4 and N_2O emissions are recalculated with global warming impact potential coefficients of 25 and 298, respectively (Eggleston et al., 2006). The uncertainty of the research results is expressed with a confidence interval (CI) at a significance level of 0.05. To characterize the uncertainty of combined study results, the combined weighted uncertainty was estimated if not stated otherwise.

Statistical analysis of the data was performed using the R Studio software at a significance level of $p < 0.05$. The Kalmogorov-Smirnov test was used to check data compliance to the normal distribution. The non-parametric method Mann-Whitney U criterion was used to compare the average values the study results. In order to take into account the relationship of affecting factors and groups of the variables, the relationship is expressed by performing a linear mixed effects regression analysis. Linear regression equations were compared with the ANCOVA test. Data with an extreme value were selected using the interquartile range as a criterion, i.e. the difference between the first and third quartiles of the data (Morillas et al., 2012). Thus, ensuring consistency with the visual representation of data dispersion in box-plot diagrams, which reflect the minimum value, first quartile, arithmetic mean (red point), median (horizontal line), third quartile, maximum value and extreme values of the sample data, while statistically significant or insignificant differences between mean values are indicated with letters like “a”, “b”, “c”. Data are plotted using the R package ggplot2, the interval (shade) around the regression line or curve indicates its 95% CI. The relationships between soil GHG emissions and affecting factors are visualized by the results of principal component analysis (PCA). Bar graphs and tables show uncertainty with a CI at a significance level of 0.05.

3. RESULTS AND DISCUSSION

3.1. Soil carbon input by foliar litter, ground vegetation and fine roots

Foliar litter. Age of the stand has the strongest correlation (*r* = 0.8) with the total biomass of tree foliar litter. While, the basal area is is more effective in predicting the annual total litter biomass depending on the tree species, however, its influence on monthly litter biomass variation and trends is not evident. The monthly variation in litter biomass in deciduous and spruce stands averages 120 ± 20% and 71 ± 21%, respectively, and the average C content in litter was 52.1 ± 0.2%. The relationship between the basal area of the forest stand and the annual litter biomass observed in the plots shows that as the basal area of spruce stands increases to 40 m^2 ha^{-1}, the annual soil C input through litter increases linearly to an average of 2.31 t C ha^{-1} $year^{-1}$. Research data indicate that as the basal area of deciduous stands increases to 10 m^2 ha^{-1}, annual litter biomass increases rapidly to an average of 1.28 t C ha^{-1} $year^{-1}$. As the basal area continues to increase, annual foliar litter biomass increases more gradually compared to spruce forests and tends to reach a plateau in deciduous forests. The average annual soil C input with litter in deciduous stands with the basal area between 11 to 46 m^2 ha^{-1} was 1.86 ± 0.46 t C ha^{-1} $year^{-1}$ during the study period (Fig. 3.1).

Another study conducted in forest stands with organic soil in Latvia show similar trends in the relationship between annual litter biomass and basal area – linear growth of litter biomass in coniferous stands over the entire range of basal area, while, in birch stands it is assumed that the amount of litter biomass is constant when reaching a basal area of 34 m^2 ha^{-1} (Bārdule et al., 2021d). However, a previous study estimated a more rapid increase in annual soil C input via litter, reaching 2.66 t C ha^{-1} $year^{-1}$ and about 3.0 t C ha^{-1} $year^{-1}$ for birch and spruce stands, respectively, as the basal area of the forest stand increased to 40 m^2 ha^{-1}. The differences may be explained by the annual variation in litterfall. To identify the factors affecting the amount of annual soil C input with foliar litter and its variations, long-term observations are needed, which would allow to evaluate the relationships between meteorological conditions and the proportions of various litter fractions with different C content. For example, annual variation in needle litterfall in a pine forest has been observed to be up to 40% (Kouki & Hokkanen, 1992). Also in a long-term study in Latvia, significant variation of annual total biomass of foliar litter was observed ranging from 2,198 to 6,085 kg ha^{-1} $year^{-1}$ (Bārdule et al., 2021a). Such variation may be explained by the impact of annual variability of meteorological conditions and extreme weather events which can significantly affect the dynamics of litterfall (Sanford et al., 1991).

Ground vegetation. Biomass of bGV (mean C content 49.7 ± 7.8%) estimated in the study sites ranges from 0.63 to 3.54 t ha^{-1} (mean 1.96 ± 0.30 t ha^{-1}). In clearcuts, bGV biomass (mean 2.24 ± 0.96 t ha^{-1}) tends to be higher than in forest stands (mean 1.91 ± 0.55 t ha^{-1}), but significant impact of the dominant tree specie, forest land status (stand or clearcut) or drainage status on the average bGV biomass was not observed.

Mean estimated aGV biomass (mean C content 47.4 ± 7.2%) in clearcut plots of 4.67 ± 0.50 t ha^{-1} (range 4.27 to 5.49 t ha^{-1}) at the end of the vegetation season was significantly higher than in forest stands – mean 1.57 ± 0.30 t ha^{-1} (from 0.39 to 3.82 t ha^{-1}). The obtained result on the aGV in forest stands is similar to which can be calculated using the aGV biomass calculation equations developed for Finnish conditions, which with a RMSE of 13.6% determine that at the age of 80 years the biomass of the aGV in forests with peat soil is 1.65 t ha^{-1}. The study of the doctoral thesis indicates that in clearcuts the biomass of aGV can be approximately three times higher than in forests. The estimated mean aGV biomass in the study clearcut plot with undrained soil is 5.08 t ha^{-1}, and in the clearcut plots with drained soil – mean 4.57 ± 0.60 t ha^{-1}.

The estimated total biomass of GV (weighted mean C content of bGV and aGV 48.2 ± 0.3%) in the study stands with drained soil ranges from 1.33 t ha^{-1} to 5.93 t ha^{-1} (mean 3.48 ± 0.60 t ha^{-1}), but in forest stands with undrained soil – from 1.46 to 6.53 t ha^{-1} (mean 3.46 ± 0.66 t ha^{-1}), respectively, in total average 3.47 ± 0.66 t ha^{-1}. In the case of clearcuts, the mean GV biomass in the plot with undrained soil (8.02 ± 1.63 t ha^{-1}) is above the maximum value of the CI of the estimated mean biomass in the clearcuts with drained soil (6.65 ± 1.02 t C ha^{-1}). In the calculation, using biomass equations developed for Finnish conditions, as well as taking into account the age structure of forests, the estimated annual soil C input in Latvia with GV biomass is 0.34 ± 0.01 t C ha^{-1} year^{-1} in birch forests and 1.29 ± 0.20 t C ha^{-1} year^{-1} in pine forests (Bārdule et al., 2021d).

Empirical data indicate a positive correlation between GV and soil nutrient availability. Of the forest stand parameters evaluated, age has the strongest correlation ($r = -0.58$, $p < 0.05$) with GV biomass. This relationship with a RMSE of ± 1.49 t ha^{-1} indicates (Fig. 3.2) that in the clearcuts the biomass of GV is on average 5.66 t ha^{-1}, which decreases to an average of 2.46 t ha^{-1} as the forest stand develops to the age of 80 years.

<u>Fine roots of trees</u>. In the study sites with drained soil, the estimated annual mortality of FR is from 0.19 ± 0.05 to 0.98 ± 0.87 t ha^{-1} year^{-1} in birch and black alder stands (mean 0.58 ± 0.44 t ha^{-1} year^{-1}), and from 0.89 ± 0.75 to 3.24 ± 2.46 t ha^{-1} year^{-1} in spruce stands (mean 1.97 ± 0.72 t ha^{-1} year^{-1}). While, in deciduous forest stands with undrained soil, the estimated annual FR mortality is from 0.69 ± 0.37 to 4.72 ± 1.15 t ha^{-1} year^{-1} (mean 2.09 ± 1.07 t ha^{-1} year^{-1}), but in spruce stand 1.09 ± 0.08 t ha^{-1} year^{-1}. The annual average mortality of FR in spruce (1.87 ± 0.66 t ha^{-1} year^{-1}) and deciduous forests (1.64 ± 0.86 t ha^{-1} year^{-1}) assessed falls within the range of uncertainty of the results of previous studies. Reported annual mortality of FR in Northern European coniferous forests is on average 2.84 ± 1.52 t ha^{-1} year^{-1}, and in deciduous forests on average 1.99 ± 1.01 t ha^{-1} year^{-1} (Neumann et al., 2020). However, the annual production of FR in birch

forests with nutrient-rich organic soil estimated in a study carried out in Estonia ranges from 1.81 to 3.02 t ha^{-1} year^{-1} (Uri et al., 2017) is higher than that estimated in the framework of the doctoral thesis research in drained birch stands (0.59 to 0.97 t ha^{-1} year^{-1}), but more in line with the estimated range of FR production in forests with undrained soil (1.43 to 3.29 t ha^{-1} year^{-1}). This may be explained by the effect of uncertainty of the empirical data. The uncertainty of average values of FR production estimated in the thesis research, depending on dominant tree specie and drainage status, is from 30 to 161% (71% on average). Another reason may be different growing conditions in the research sites, as empirical data indicate that the annual production of fine roots tends to be higher as the average GW level and soil nutrient availability decrease. Higher FR production in nutrient-poorer soil has also been observed in other studies (Leppälammi-Kujansuu et al., 2014; Lehtonen et al., 2016; Mäkelä et al., 2016; Kriiska et al., 2019). The increased production of FR in the study sites with undrained soil could be promoted by the dynamics of the monthly GW level. The average GW level in the study sites with undrained soil during the summer months, when root growth can be greatest (Varik et al., 2015), was deeper than 40 cm, which has also been the case in the mentioned Estonian study with drained soil.

Among the evaluated characteristics of the forest stand, the basal area has the strongest correlation with annual mortality of FR (r = 0.30). The obtained data indicate that as the basal area of the forest stands increases from 10 to 40 m^2 ha^{-1}, the annual mortality of FR biomass increases from an average of 0.64 to 0.99 t ha^{-1} year^{-1} (Fig. 3.3). However, the regression equation constructed according to the empirical data obtained in the study has a high RMSE of ± 1.43 t ha^{-1} year^{-1}, or 81% of the average value of annual FR mortality estimated in the study sites.

3.2. Soil greenhouse gas emission estimation factors and equations

<u>Soil CH_4 emissions</u>. The average coefficient of variation of monthly CH_4 measurement results at study sites with drained soil is 60%, while at sites with undrained soil it is 268%. The mean value of monthly CH_4 emissions measured at study sites with drained soil ranges from –7.15 ± 2.86 to 2.87 ± 14.04 kg CH_4C ha^{-1} year^{-1}, and at sites with undrained soil it ranges from –4.56 ± 2.35 to 497.15 ± 1,558.67 kg CH_4-C ha^{-1} year^{-1}.

Monthly measurements of soil CH_4 emissions show high variation. Monthly soil CH_4 emissions below –12.26 kg CH_4-C ha^{-1} and above 5.61 kg CH_4-C ha^{-1} were identified as extreme. Extreme values were observed in three sites with undrained soil (mean value of extreme emissions 877.76 ± 1,424.652 kg CH_4-C ha^{-1} year^{-1}) and four sites with drained soil (mean value of extreme emissions 27.53 ± 23.48 kg CH_4-C ha^{-1} year^{-1}). In one of the black alder stands with undrained soil, excessive high values of extreme emissions (mean 1,355.81 ± 1,682.84 kg CH_4-C ha^{-1} year^{-1}) reaching up to 4,933.09 ± 2,5517.45 kg CH_4-C ha^{-1} year^{-1} in nine sampling occasions were observed. A similar observation was made in a study of the northern peatlands, where three episodes of extremely high CH_4 emissions resulted in 1,020 kg CH_4-C ha^{-1} flux from the soil to the atmosphere (Glaser et al., 2004). In the other nine sites with undrained soil, the mean value of extreme emissions (17.27 ± 9.3 kg CH_4-C ha^{-1} $year^{-1}$) was lower than in sites with drained soil. The results suggest that highly extreme soil CH_4 emissions, which can significantly impact the mean emissions, can be expected in about 10% of forest areas with undrained nutrient-rich organic soil. Furthermore, the trend of higher CH_4 emissions is observed in stands dominated by the black alder. Another study assessed the spatial heterogeneity of soil CH_4 emissions in the northern peatlands, finding that 10% of the area with a GW level close to the soil surface (under conditions of soil saturation) can produce up to 45% of total CH_4 emissions (Sachs et al., 2011).

The study estimates annual soil CH_4 emissions (Table 3.1) to be between –8.2 to 15.3 kg CH_4-C ha^{-1} $year^{-1}$ in sites with drained soil, with a mean value of –3.47 ± 0.94 kg CH_4-C ha^{-1} $year^{-1}$. In sites with undrained soil, the estimated emissions are between –6.5 to 1,016.2 kg CH_4-C ha^{-1} $year^{-1}$, with a mean value of 106.6 ± 101.0 kg CH_4-C ha^{-1} $year^{-1}$. The black alder stands have higher emissions, with a mean of 199.8 ± 393.2 kg CH_4-C ha^{-1} $year^{-1}$ in undrained soil and 6.8 ± 16.6 kg CH_4-C ha^{-1} $year^{-1}$ in drained soil. Birch and spruce forest stands have negative emissions, with a mean of –4.4 ± 1.2 kg CH_4-C ha^{-1} $year^{-1}$ for both drained and undrained soil. The clearcut study sites have estimated emissions ranging from –6.0 to 6.88 kg CH_4-C ha^{-1} $year^{-1}$, with a mean value of –2.4 ± 4.6 kg CH_4-C ha^{-1} $year^{-1}$. The range of emissions from drained organic soil estimated is similar to those estimated in Finland – from –2.8 kg CH_4-C ha^{-1} $year^{-1}$ in nutrient-poor soils to 11.6 kg CH_4-C ha^{-1} $year^{-1}$ in nutrient-rich soil (Ojanen et al., 2013). While, the estimated mean annual CH_4 emissions from undrained soil are within the range specified by the IPCC guideline (0 to 493 kg CH_4-C ha^{-1} $year^{-1}$, with a mean of 137 kg CH_4-C ha^{-1} $year^{-1}$) for nutrient-rich organic soil with a restored natural moisture regime in the boreal forest (Hiraishi et al., 2014).

There is a strong correlation ($r = 0.8$) between monthly GW level and CH_4 emission measurements. However, the uncertainty of the emissions is not uniform and is greater when the GW level is shallower than 20 cm. Average CH_4 emissions estimated in the study were 87.5 ± 97.3 kg CH_4-C ha^{-1} $year^{-1}$ when the groundwater was shallower and –4.4 ± 0.2 kg CH_4-C ha^{-1} $year^{-1}$ when it was deeper (Fig. 3.4). Previous research in peatlands and bogs of the temperate and boreal zone shows that significant CH_4 emissions are expected when the water level is shallower than 20 cm (Couwenberg & Fritz, 2012). Taking into account the uncertainty of the data, the results of the doctoral thesis and previous research are comparable – in the boreal zone, the estimated CH_4 emissions from peatlands with GW level shallower and deeper than 20 cm from the topsoil are, respectively, from –1.7 to 525 kg CH_4 ha^{-1} $year^{-1}$ (on average 56 kg CH_4 ha^{-1} $year^{-1}$) and –1.1 to 51 kg CH_4 ha^{-1} $year^{-1}$ (mean 8.6 kg CH_4 ha^{-1} $year^{-1}$) (Couwenberg & Fritz, 2012). The findings are in line with the known observation that a 20 cm soil layer with aerobic conditions can be sufficient to oxidize most of the CH_4 produced in the anaerobic conditions before it enters the atmosphere (Hornibrook et al., 2009).

The mean value of GW level measurements in the study sites has moderate correlation ($r = -0.64$) and strong correlation ($r = -0.88$) with calculated annual total soil CH_4 emissions, taking into account and excluding the study site with an extreme annual cumulative emission value, respectively. The GW level affects the proportion of CH_4-producing (methanogens) and CH_4-consuming microorganisms (methanotrophs), which regulate the balance between CH_4 emissions and oxidation (Couwenberg & Fritz, 2012). The average GW level can accurately indicate annual soil CH_4 emissions, because the methanogens and methanotrophic microorganisms are well adapted to the stress of adverse conditions and remain abundant at a certain depth below the soil surface, regardless of the GW level fluctuations (Kettunen et al., 1999; Knorr & Blodau, 2009; Kip et al., 2012). According to the results of the annual cumulative emissions and the average depth of the GW level obtained in the study, two depth ranges of the GW level with a limit value of 31 cm can be distinguished. For the GW level deeper than 31 cm, the lines of the linear regression plot overlap regardless of whether the statistically extreme annual emission value is considered in the analysis. The extreme value of annual CH_4 emissions (1,036.7 ± 834.4 kg CH_4-C ha^{-1} $year^{-1}$) when the GW level is shallower than 31 cm significantly affects the slope coefficient of the linear regression equation (Fig. 3.5). These ranges correspond to the IPCC-defined threshold of 30 cm separating shallowly or deeply drained soils (Hiraishi et al., 2014).

Soil N_2O emissions. The mean measured instantenious N_2O emissions from drained soil ranges from 0.6 ± 0.6 kg N_2O-N ha^{-1} $year^{-1}$ in black alder stands to 1.5 ± 1.3 kg N_2O-N ha^{-1} $year^{-1}$ in clearcuts. On the other hand, the average N_2O emissions from undrained soil range from 0.0 ± 0.1 to 3.3 ± 4.0 kg N_2O-N ha^{-1} $year^{-1}$ in clearcut plots and black alder stands, respectively. In birch, black alder, and clearcut stands, the soil drainage status has a significant impact on the average N_2O measurement result. However, in spruce stands, the average N_2O emissions do not significantly differ between drained and undrained soil. The average N_2O emissions from drained soil in birch, black alder, and spruce stands are 0.842 ± 0.33, 0.615 ± 0.54, and 1.092 ± 0.60 kg N_2O-N ha^{-1}, respectively, and are not significantly different from each other. However, in study sites with undrained soil, the mean N_2O emissions from birch, spruce, and black alder stands (2.85 ± 1.46; 0.64 ± 0.33; and 3.31 ± 1.52 kg N_2O-N ha^{-1} $year^{-1}$, respectively) are significantly different from each other. The average N_2O emissions from undrained soil and drained soil in the study sites are 2.6 ± 0.9 kg N_2O-N ha^{-1} $year^{-1}$ and 1.1 ± 0.4 kg N_2O-N ha^{-1} $year^{-1}$, respectively. The difference between these values is significant ($p = 0.01$) (Fig. 3.6).

When excluding extreme values of soil N_2O emissions, changes in *Ts* explain 44% of the variation in emissions. The correlation between *Ts* and emissions from drained soil is moderate ($r = 0.48$), but weak for emissions from undrained soil. When all data is used in the correlation analysis, both *Ts* and groundwater level measurements have a weak correlation ($r = 0.3$) with soil N_2O emissions.

The average annual N_2O emissions of drained nutrient-rich organic forest soil in the study, estimated at 1.1 ± 0.4 kg N_2O-N ha^{-1} $year^{-1}$, are lower than the IPCC default EF of 3.2 kg N_2O-N ha^{-1} $year^{-1}$ for drained nutrient-rich organic soil in the boreal zone. However, it falls within the 95% CI of the default EF for the temperate zone, which is 2.8 kg N_2O-N ha^{-1} $year^{-1}$ with a CI of –0.57 to 6.1 kg N_2O-N ha^{-1} $year^{-1}$ (Hiraishi et al., 2014). The estimated annual average N_2O emissions from drained organic soils in Finland, based on their nutrient content, range from 0.18 ± 0.04 to 2.11 ± 0.64 kg N_2O-N ha^{-1} $year^{-1}$, according to Statistics Finland (2014). The IPCC guidelines consider N_2O emissions from organic soils with a restored natural moisture regime to be negligible. However, the average annual emissions estimated in this study from undrained soils (2.6 ± 0.9 kg N_2O-N ha^{-1} $year^{-1}$) were found to be higher than those from drained soils, as shown in Table 3.2.

Soil respiration CO_2 emissions. The average values of *Rtot* measurements taken in the study stands with different dominant tree species do not differ significantly from one another ($p > 0.05$). The mean values of *Rtot* measurements (Fig. 3.7) are also not significantly different between the different forest types ($p > 0.05$) or soil drainage status ($p = 0.34$). Accordingly, the mean estimated *Rtot* in the study sites with drained soil and undrained soil are 7.35 ± 0.89 and 7.02 ± 0.96 t CO_2-C ha^{-1} $year^{-1}$, respectively, and these values are not significantly different ($p = 0.34$). The average depth of the GW level in the study sites with drained soil was 55 ± 2 cm, while in the sites with undrained soil it was 35 ± 3 cm, with a mean difference of 18 ± 2 cm. However, this difference is not reflected in the *Rtot* measurements, as the measurement of the GW and *Rtot* results have a weak correlation ($r = 0.3$). The average values of *Rtot* measurements are significantly different ($p = 0.002$) between forest stands (6.84 ± 0.56 t CO_2-C ha^{-1} $year^{-1}$) and clearcuts (10.08 ± 1.96 t CO_2-C ha^{-1} $year^{-1}$). It is likely that soil damage from mechanized logging and decomposition of logging residues contribute to higher *Rtot* emissions in clearcuts as reported by James & Harrison (2016) and Jandl et al. (2007). Similar trends have been observed in a Scottish bog, where the estimated annual *Rtot* is 4.53 t CO_2-C ha^{-1} $year^{-1}$ in drained and undrained forested areas, and 6.95 t CO_2-C ha^{-1} $year^{-1}$ in areas without forest cover (Yamulki et al., 2013).

The variation of *Rtot* measurement results is primarily driven by changes in atmospheric and, as a result, soil temperatures. There is a strong correlation between *Rtot* and *Ts* measurement results ($r = 0.89$). The relationship between *Rtot* and *T*s can be described by an exponential regression equation (as shown in Fig. 3.8), which shows that as *Ts* increases from –1.0 to 22.0°C, *Rtot* increases from 1.5 to 29.2 t CO_2-C ha^{-1} $year^{-1}$. The study found that the depth of the groundwater level had a weak influence ($r = 0.30$) on *Rtot*.

The *Rtot* emissions in clearcuts are higher than those in forest stands across the range of soil temperatures studied. As the *Ta* increases from 0 to 20°C, the difference in *Rtot* CO_2 emissions between clearcuts and stands increases from a mean of 1.5 to 15.6 t CO_2-C ha^{-1} $year^{-1}$. Based on the relationship between *Ta* and CO_2 emission measurements (as shown in Fig. 3.8), if the soil temperature is at 7°C, which corresponds to the average annual air temperature in Latvia according to LVĢMC, the predicted CO_2 emissions from *Rtot* in clearcuts and forest stands would be 4.9

and 3.7 t CO_2-C ha^{-1} $year^{-1}$ respectively, with a difference of 1.2 t CO_2-C ha^{-1} $year^{-1}$ and a RMSE of ± 3.67 and ± 2.4 t CO_2-C ha^{-1} $year^{-1}$, respectively.

According to data from meteorological stations in Latvia from 2012 to 2021, the estimated average annual *Rtot* emissions in clearcuts range from 4.5 to 11.4 t CO_2-C ha^{-1}, with an average of 7.70 ± 0.53 t CO_2-C ha^{-1} $year^{-1}$. In forest stands, the estimated average annual *Rtot* emissions range from 3.8 to 9.9 t CO_2-C ha^{-1}, with an average of 6.14 ± 0.15 t CO_2-C ha^{-1} $year^{-1}$. Despite the fact that the average *Rtot* measurements in sites with different dominant tree species and soil drainage status do not significantly differ, the calculation of annual emissions shows that both soil drainage status and dominant tree species have a significant impact on average annual *Rtot* CO_2 emissions over a 10-year period (Fig. 3.9). The data used in the modelling shows that the annual variation in air temperature can affect the cumulative *Rtot* emissions annually by 0.3 to 3.3 t CO_2-C ha^{-1} (with an average of 1.6 t CO_2-C ha^{-1}). This highlights the importance of considering the historical and projected dynamics of air temperature when predicting soil emissions at a national level and in the long term. Due to the variation of air temperature from 2012 to 2021, the coefficient of variation of modelled annual *Rtot* CO_2 emissions at individual sites ranges from 1.2% to 13.4% with an average of 8.2%. The average annual *Rtot* CO_2 emissions over the study year and the following 10 years were found to be not significantly different (Table 3.3), which can be attributed to the air temperature dynamics in the study year being representative of the climate. This approach for estimating annual *Rtot* CO_2 emissions has relatively low uncertainty compared to the Finnish National GHG Inventory, which uses a CO_2 emission factor with an uncertainty of 150% (Statistics Finland, 2014).

The estimated mean annual *Rhet* of 3.80 ± 0.44 t CO_2-C ha^{-1} for drained soil (ranging from 2.9 to 4.4 t CO_2-C ha^{-1} in study stands) is consistent with other *Rhet* estimates found in similar studies in the region. *Rhet* by direct measurement methods has been most extensively studied in Finnish forests with the following *Rhet* results: 1.85 ± 0.09 to 4.26 ± 0.26 t CO_2-C ha^{-1} from drained organic soils with varying nutrient availability (Minkkinen et al., 2007); 1.46 to 6.70 t CO_2-C ha^{-1} $year^{-1}$ from drained peat (Ojanen et al., 2010); 2.07 to 5.39 t CO_2-C ha^{-1} $year^{-1}$ from the organic soil of afforested cropland and 2.76 to 4.79 t CO_2-C ha^{-1} $year^{-1}$ from peat in a recultivated peat extraction site (Mäkiranta et al., 2007). Another study covering a region from Estonia to Finland estimated the *Rhet* of drained forest peat between 2.48 and 5.15 t CO_2-C ha^{-1} $year^{-1}$ (Minkkinen et al., 2007).

Soil net CO_2 emissions. According to the empirical data collected in the study, soil was the net source of CO_2 in the clearcuts with undrained (0.49 ± 1.02 t CO_2-C ha^{-1} $year^{-1}$) and drained area (0.89 ± 0.99 t CO_2-C ha^{-1} $year^{-1}$), as well as in drained birch stands (0.50 ± 1.08 t CO_2-C ha^{-1} $year^{-1}$). Undrained and drained soil in deciduous stands were a net CO_2 sink of mean of 0.32 ± 0.93 t CO_2-C ha^{-1} $year^{-1}$ and 0.94 ± 1.38 t CO_2-C ha^{-1} $year^{-1}$, respectively, while in spruce stands net removals of 0.88 ± 1.01 t CO_2-C ha^{-1} $year^{-1}$ and 0.60 ± 74 t CO_2-C ha^{-1} $year^{-1}$, are estimated, respectively (Fig. 3.10).

In forested areas, both GV and foliar litter are significant sources of annual soil C input. In the study year, GV and foliar litter in stands with different dominant tree species provided an average of 1.72 ± 0.33 t C ha^{-1} $year^{-1}$ and 1.79 ± 0.25 t C ha^{-1} $year^{-1}$, or 41 ± 8% and 43 ± 6% of the total soil C input, respectively. The contribution of FR to the total soil C input was estimated to be 0.71 ± 0.34 t C ha^{-1} $year^{-1}$, or 16 ± 7% of the total input. In clearcuts, the lack of soil C input from FR and litterfall is typically compensated by higher GV biomass. The total estimated soil C input from GV in forest stands was 3.47 ± 0.54 t C ha^{-1} $year^{-1}$, while in clearcuts, the estimated GV provided an average of 6.92 ± 0.96 t C ha^{-1} $year^{-1}$, which is approximately two times higher than in forested areas.

The assessment of the study – the soil in forest stands is a net C sink – is in accordance with the results of the previous study conducted in Latvia on changes in soil C stock in forestry drained peatlands (Lupikis & Lazdins, 2017). This can be explained by the soil C input by biomass mortality, which is fully capable of compensating the annual C losses caused by soil respiration. In the study, the soil in the clearcuts was evaluated as a source of CO_2 emissions and the CO_2 emissions caused by soil respiration were higher but the annual C input was lower compared to forest stands. Although C sequstrated by GV in clearcuts (on average 3.55 ± 0.37 t C ha^{-1} $year^{-1}$) was significantly higher than in forests (1.65 ± 0.37 t C ha^{-1} $year^{-1}$), it could not fully compensate for by mean 0.8 t CO_2-C ha^{-1} $year^{-1}$ higher soil CO_2 emissions and absence of soil C inputs with foliar litter (in forest stands mean 1.8 ± 0.5 t C ha^{-1} $year^{-1}$) and FR (in forest stands mean 0.71 ± 0.37 t C ha^{-1} $year^{-1}$).

3.3. Soil greenhouse gas emission factors

The results of the PCA of the mean values of measurement in the study sites (as shown in Figure 3.11) indicate that a lower average groundwater level is associated with higher C concentration and C/N ratio in the topsoil. This correlation is confirmed by the correlation analysis (r = 0.5, p < 0.05) shown in Figure 3.12. The relationship between the C/N ratio and the depth of the GW suggests that a long-term low GW increases the degree of peat mineralization but reduces the mineralization activity. This is reflected in the significant negative correlation between soil GHG emissions and the depth of the GW, with the greatest reduction in net CH_4 emissions observed under low groundwater levels (significant negative correlation with CH_4, ρ = –0.9, p < 0.05 and N_2O, ρ = –0.4, p < 0.05).

According to PCA, a lower average level of GW and a thicker peat layer are associated with a higher annual production of FR, as a result, higher FR mortality. Root production has a negative relationship with bGV biomass and soil nutrient availability indicators, such as K, Ca, Mg and P. Correlation analysis confirms that FR mortality has a significant negative correlation with bGV (ρ = –0.6; p < 0.05) and soil K concentration (Spearman and Pearson correlation coefficient −0.6; p < 0.05). This aligns with the findings of a previous study (Lehtonen et al., 2016) that showed that trees tend to compensate for nutrient unavailability with higher root biomass. At the same time, PCA indicates that soil nutrient availability has a direct relationship with soil emissions, most significantly affecting CH_4 emissions and least affecting soil respiration (CO_2 emissions). The relationship between soil nutrient status and GHG emissions is confirmed by Spearman's correlation analysis. Specifically, CH_4 emissions are significantly correlated with soil Ca ($r = 0.5$, $p < 0.05$) and Mg ($r = 0.6$, $p < 0.05$) concentrations, while N_2O emissions are significantly correlated with C ($r = 0.5$; $p < 0.05$) and N ($r = 0.6$; $p < 0.05$) concentration, but *Rtot* is significantly correlated with soil C concentration ($r = 0.5$; $p < 0.05$). It is significant that both nutrient availability and higher soil CH_4 emissions are influenced by higher soil pH values, as indicated by both PCA and correlation analysis. Soil acidity is known to affect the population of methanogens and methanotrophs (Serrano-Silva et al., 2014). In addition, the higher availability of macroelements K, Ca, Mg, as well as P is also reflected in larger bGV biomass (Fig. 3.11). Biomass of bGV is significantly correlated with concentrations of N and K ($r = 0.5$; $p < 0.05$), as well as P ($r = 0.7$; $p < 0.05$) and C/N ratio ($r = -0.6$; $p < 0.05$) in soil. PCA also indicates that, of the assessed stand characteristics, stand age has the strongest relationship with annual foliar litter biomass. This is also confirmed by the correlation analysis, the highest correlation coefficient ($r = 0.8$; $p < 0.05$) was found for the relationship with the age of the forest stand. According to correlation analysis, stand characteristics do not correlate with GHG emissions, but PCA indicates that stand age has an inverse relationship with *Rtot*. Namely, soil respiration tends to decrease as the development of the forest stand continues and its age increases (Fig. 3.11). This is partially explained by the significant negative Pearson correlation of stand age with bGV ($r = -0.4$; $p < 0.05$) and aGV ($r = -0.6$; $p < 0.05$). Both PCA and Pearson correlation analysis ($r = 0.7$; $p < 0.05$) indicate that of the GV components, its aboveground biomass has the greatest influence on autotrophic respiration.

3.4. Ecosystem greenhouse gas emissions

The estimated annual mean GHG removals of forests with undrained soil and dominant tree specie of birch and spruce is 1.8 ± 7.57 t CO_2 eq. ha^{-1} $year^{-1}$ and 2.8 ± 8.3 t CO_2 eq. ha^{-1} $year^{-1}$, respectively, while the estimated annual mean GHG emissions of black alder forests with undrained soil are 3.3 ± 13.6 t CO_2 eq. ha^{-1} $year^{-1}$. Black alder forests are estimated as a source of net GHG emissions mainly under the influence of the empirical data of soil CH_4 emissions obtained in the study (Table 3.1). Empirical data indicate that annual CH_4 emissions from undrained soils of black alder forests are mean 4.4 ± 3.1 t CO_2 eq. ha^{-1} $year^{-1}$, while in other groups of research objects the annual estimated CH_4 emissions are relatively insignificant (Fig. 3.13). Although extreme soil CH_4 emissions were detected in one of the five plots located in black alder forests with undrained soil, the probability of occurrence of such emissions in forests with undrained soil cannot be ignored. The estimated annual mean GHG removals of forests with drained soils and dominant tree species spruce and black alder is 4.6 ± 12.8 t CO_2 eq. ha^{-1} $year^{-1}$ and 4.2 ± 17.7 t CO_2 eq. ha^{-1} $year^{-1}$, respectively. It is estimated that birch stands with drained soil tend to be climate neutral, with average annual GHG emissions of 0.0 ± 11.5 t CO_2 eq. ha^{-1} $year^{-1}$.

The long-term cumulative effect of annual GHG emission dynamics on forest ecosystem GHG emissions can be seen in Figure 3.14. The results obtained in the study indicate that, on average, the forest ecosystem of all study object groups (except for black alder forests with undrained soil) is a net sink of GHG emissions in the long term. However, uncertainty must be taken into account when interpreting estimated average annual or long-term cumulative ecosystem GHG emissions or removals. Long-term cumulative emissions of black alder forests and birch forests with drained soil, as shown in Figure 3.14, should be interpreted with particular caution. Taking into account the uncertainty of empirical data, in the long term, the black alder forest can be both an annual source and sink of GHG emissions (Fig. 3.13), but the climate neutrality result of birch forests with drained soil was obtained using empirical data with a combined uncertainty of 134% in the calculation. The trends of cumulative GHG emissions of spruce and birch forests with undrained soil indicate that the climate neutrality assessment of birch forests with drained soil can also be a cumulative effect of the uncertainty of the calculation components. Also, the calculation assumptions about the course of growth of forest stands and the intensity of harvesting can have a significant impact on the obtained calculation result of net GHG emissions of the forest ecosystem.

Taking into account annual soil CO_2 emissions and soil C stock dynamics during forest management in the long term, calculated according to the empirical data obtained in the study, drained and undrained nutrient-rich organic forest soil annually sequester on average 0.28 ± 0.66 t C ha^{-1} $year^{-1}$ and 0.42 ± 0.43 t C ha^{-1} $year^{-1}$, respectively. In birch, spruce and black alder forest forests, the undrained soil sequestrates an average of 0.64 ± 0.51 t C ha^{-1} $year^{-1}$; 0.30 ± 0.33 t C ha^{-1} $year^{-1}$ and 0.33 ± 0.33 t C ha^{-1} $year^{-1}$ but drained soil -0.34 ± 0.26 t C ha^{-1} $year^{-1}$, 0.35 ± 0.54 t C ha^{-1} $year^{-1}$ and 0.86 ± 0.53 t C ha^{-1} $year^{-1}$, respectively (Fig. 3.15). During the forest management cycle, soil C stock can both increase and decrease annually. The data collected in the study indicate that it is determined by the development stage of the forest stand. During the period of clearcut, the forest soil loses C, but as the forest stand develops, it becomes a C sink. The increase in soil C sequestration is mainly determined by the uptake of soil C by foliar litter and FR, which tend to increase with increasing age of the forest stand.

Soil CO_2 emissions in Finland were estimated to increase with soil nutrient availability, from 3.8 to 12.10 t C ha^{-1} $year^{-1}$ (Ojanen et al., 2010). Despite this, doctoral research showed that nutrient-rich organic forest soils can still be net CO_2 sinks, which aligns with previous studies indicating that C stock in boreal forests may remain unchanged or increase after drainage of nutrient-rich organic soil (Meyer et al., 2013; Varik et al., 2015). An important aspect that can affect the conclusions of various studies is the methodology used to assess the dynamics of soil C stock, which may or may not take into account different components of soil C input (Ojanen et al., 2012).

According to the empirical data collected in the study and the methodology of forest ecosystem GHG emission calculations, during the 240-year forest land management cycle, forest ecosystems with naturally wet nutrient-rich organic soil are a net GHG sink of mean 0.2 ± 9.7 t CO_2 eq. ha^{-1} $year^{-1}$, but a forest ecosystem with drained nutrient-rich organic soil a net sink of mean 2.9 ± 14.4 t CO_2 eq. ha^{-1} $year^{-1}$. The dispersion of annual GHG emission values of forest ecosystems with drained or undrained soil is significantly different (Fig. 3.16). Thus, the results indicate that forests with dried soil can provide a greater contribution to mitigating climate change.

CONCLUSIONS

1. A linear relationship was found between the basal area and annual soil carbon input by foliar litter in spruce stands ($r = 0.9$). In birch and black alder stands, the annual soil C input increases rapidly till the basal area reaches about 10 m^2 ha^{-1}, resulting in similar levels of soil C input as in spruce stands with a basal area of around 30 m^2 ha^{-1}. Therefore, deciduous forests with smaller basal areas have the potential to provide a greater soil C input by foliar litter than spruce stands and, in managed forests, could potentially contribute more to the preservation of soil C stock.

2. The ground vegetation has a crucial role in maintaining soil carbon stock in clearcuts by potentially compensating for the lack of C input by litterfall and tree fine roots. Soil C input by the ground vegetation in clearcuts is significantly higher than in stands ($p < 0.05$), with values of 3.3 ± 0.5 and 1.7 ± 0.3 t C ha^{-1} $year^{-1}$, respectively. Additionally, the relationship between ground vegetation biomass and stand age ($r = -0.6$) shows that ground vegetation biomass in clearcuts is approximately twice as large as in 80-year-old stands.

3. The study did not find a significant relationship between stand age, tree diameter, or growing stock and the annual mortality of tree fine roots (mean 1.5 ± 0.8 t ha^{-1} $year^{-1}$).

4. Analysis of the relationship between annual soil CH_4 emissions and average groundwater level ($r = -0.6$) revealed that if the mean groundwater level depth below ground surface is less than 30 cm, the soil becomes the source of CH_4 emissions. While relationships found between groundwater level and soil CH_4 emissions measurements are similar in both drained and undrained sites, the higher probability of significantly increased emissions in undrained sites highlights the importance of assessing the functionality of the drainage system when estimating emissions.

5. The study found a moderate negative correlation ($r = -0.4$) between the mean values of groundwater level measurements and the estimated annual total soil N_2O emissions. Furthermore, analysis revealed a significant difference ($p < 0.01$) between the annual total soil N_2O emissions in drained sites (mean 1.1 ± 0.4 kg N ha^{-1} $year^{-1}$) and undrained sites (mean 2.6 ± 0.9 kg N ha^{-1} $year^{-1}$).

6. In Latvian climatic conditions, the estimated annual CO_2 emissions by total soil respiration were higher in clearcuts (mean 7.7 ± 0.5 t C ha^{-1} $year^{-1}$) than in forest stands (mean 6.1 ± 0.2 t C ha^{-1} $year^{-1}$) with statistical significance ($p < 0.05$). There was no significant effect found of drainage or the dominant tree species on instantenious total soil respiration CO_2 emissions.

7. During the forest management cycle, soil carbon stock losses in nutrient-rich drained and undrained organic forest soil in clearcuts (mean 0.7 t C ha^{-1} $year^{-1}$) are offset by soil carbon sequestration in stands (mean of 0.6 t C ha^{-1} $year^{-1}$).

8. Managed forests with drained soil have the potential to make a greater contribution to climate change mitigation, as forests with drained and un-drained nutrient-rich organic soils can sequester, on average, 2.9 and 0.2 t CO_2 eq. $year^{-1}$, respectively.

INFORMĀCIJAS AVOTU SARAKSTS / *REFERENCES*

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GHG Emissions from Drainage Ditches in Peat Extraction Sites and Peatland Forests in Hemiboreal Latvia

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Abstract: We determined the magnitude of instantaneous greenhouse gas (GHG) emissions from drainage ditches in hemiboreal peatlands in Latvia during the frost-free period of 2021 and evaluated the main affecting factors. In total, 10 research sites were established in drained peatlands in Latvia, including active and abandoned peat extraction sites and peatland forests. Results demonstrated that in terms of global warming potential, the contribution of CO_2 emissions to the total budget of GHG emissions from drainage ditches can exceed the CH_4 contribution. The average CO_2 and N_2O emissions from drainage ditches in peatland forests were significantly higher than those from ditches in peat extraction sites, while there was no difference in average CH_4 emissions from ditches between peatland forests and peat extraction sites. Emissions from ditches of all GHGs increased with increasing temperature. In addition, CO_2 and N_2O emissions from drainage ditches increased with decreasing groundwater (GW) level. They were also negatively correlated with water level in ditches, but positively with potassium (K) and total nitrogen (TN) concentrations in water. By contrast, CH_4 emissions from drainage ditches increased with increasing GW level and water level in ditches but were negatively correlated with K and TN concentrations in water.

Keywords: greenhouse gases; carbon dioxide; methane; nitrous oxide; drainage ditches; emissions; peatland forests; peat extraction sites

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1. Introduction

Drainage ditch networks are common man-made elements of many landscapes with peat (organic) soils and are generally dug to lower GW levels for peat drying and subsequent extraction and/or to improve agricultural and forest productivity [1–5]. Establishment and maintenance of drainage systems ensure sufficient aeration of upper soil layers to support development and growth of vegetation, including trees [1], but simultaneously cause soil disturbances, which alter GHG emissions and removals at the landscape level [6,7]. In Latvia, drainage of agricultural lands began to be extended at the end of the 16th century and the start of the 17th century, but drainage of forest land started only in the first half of the 19th century [8]. The first records of peat extraction in Latvia date back to the second half of the 17th century and the early 18th century [9]. Currently, in Latvia, drained organic soils comprise 425.1 kha in forest land, 76.0 kha in grassland, 78.6 kha in cropland, 39.7 kha in wetlands (peat extraction fields), and 9.3 kha in settlements (628.6 kha in total) [10].

As drained peatlands in general contribute significantly to the global anthropogenic GHG emissions [11], peatland management has received much attention, especially during the past several years (e.g., [12]) in the context of ambitious aims to achieve carbon (C) neutrality by 2050–2070 under the Paris Agreement [13]. GHG emissions not only from the drained soils but also from drainage ditches themselves, especially eutrophic ditches with organic-rich sediment, can appreciably contribute to the total GHG budgets of drained areas [14–17]. Emissions from ditches, which are anthropogenic in origin, cannot therefore be ignored when landscape- or national-scale GHG budgets are estimated [5–7,18]. The latest Intergovernmental Panel on Climate Change (IPCC) guidelines for national GHG inventories also provide methodologies and emission factors for CH_4 emissions from drainage ditches, while methodologies and emission factors for CO_2 and N_2O emissions from drainage ditches have until now not been provided [6,19].

Carbon dioxide (CO_2) is produced by the respiration of both plants and soil microorganisms and by other biological processes in sediments [16,20]. During the daytime, CO_2 emissions may decrease due to CO_2 uptake by photosynthetically active aquatic plants [20]. Although CO_2 is highly soluble in water, oversaturation of CO_2 near the sediment/water interface can release CO_2 to the atmosphere [16]. The magnitude of methane (CH_4) emissions depends on the dominance of two counteracting microbial processes: methanogenesis, which is the production of CH_4 either by acetate fermentation or by CO_2 reduction in anoxic conditions (the terminal microbial process of organic matter degradation), and the following oxidation of the generated CH_4 into CO_2 by methanotrophic bacteria [16,21]. Nitrous oxide (N_2O) emissions result from biogeochemical interactions between reactive nitrogen (N), microorganisms (nitrification and denitrification processes), aquatic plants, and the environment, and such emissions help to identify drainage ditches, especially those suffering from eutrophication, as sources of N_2O [22–25]. In general, GHG is transported to the atmosphere through water by three main pathways: (1) diffusion between soil and atmosphere (2) bubble ebullition, and (3) plant-mediated transport [20,26,27].

GHG production is driven by biochemical processes (microbial processes being the key processes) and emissions as the terminal process is regulated by variables such as the trophic state of the water body; sediment texture and chemistry, including organic matter availability; water chemistry, including pH and electrical conductivity (EC), oxygen (O_2) saturation and the presence of electron acceptors such as O_2, NO_3^-, Fe^{3+}, and SO_4^{2-} (redox conditions); and sediment and water temperature (e.g., [2,16,25,28–32]). In addition, GHG emissions from drainage ditches vary depending on factors such as water level and flow rate in a ditch, frequency and duration of drought, water body morphology, plant community composition, and dominant land use in the catchment (e.g., [5,7,17,33]).

Our objective in this study was to investigate the magnitude of GHG emissions (CO_2, CH_4 and N_2O exchange at the water/air interface) from drainage ditches in hemiboreal peatlands in Latvia and to identify the main affecting factors. As research sites were located both in peat extraction sites (active peat extraction sites, abandoned peat extraction sites with bare peat and with shrub and herbaceous plant vegetation) and in peatland forests (dominated by Scots pine and silver birch), the results allowed for indirect assessment of the potential impact of afforestation of peat extraction sites on GHG emissions from drainage ditches.

2. Materials and Methods

2.1. Research Sites

This study was conducted in ten research sites in drained hemiboreal peatlands (former and active peat extraction fields) in Latvia covering different regions (Figure 1) during the frost-free period of 2021. In 2021, the weather conditions in Latvia were typical (representative) for the region and no significant deviations from the norm were detected. In 2021, the mean annual precipitation in Latvia was 676.3 mm, and it was 1% below the annual norm (685.6 mm). Thus, 2021 was already the 4th consecutive year with less than usual precipitation. The mean annual air temperature was 7.0 °C, the minimum mean monthly temperature was −5.2 °C (February 2021), and the maximum mean monthly temperature was 21.5 °C (July 2021). In 2021, the average air temperature was 0.2 °C warmer than the climatic standard norm (1991–2020), and thus, 2021 was already the 9th consecutive year warmer than the climatic standard norm [34].

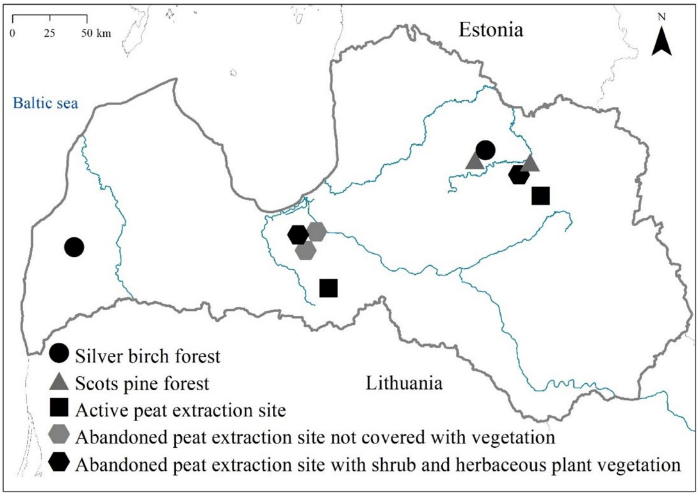

Figure 1. Location of research sites in Latvia.

One representative ditch was selected at each research site (Table 1). Research sites represent and were divided into five different groups according to the type of current land use (two research sites in each group): (i) active peat extraction sites; (ii) abandoned peat extraction sites not covered with vegetation (bare peat); (iii) abandoned peat extraction sites with shrub and herbaceous plant vegetation; (iv) Scots pine forest with organic soil; and (v) silver birch forests with organic soil (Table 1). All research sites are former peat extraction fields, with the difference that four sites have been afforested following peat extraction (Table A1), four sites are abandoned, and two sites are still under active peat extraction. In all research sites, current management practice has continued for at least 20 years.

Table 1. Characterization of the research sites in Latvia.

Current Land Use/Type of Vegetation	Research Site	Coordinates (LKS92 TM Coordinate System)	Ditch width at the Soil Surface Level, cm	Ditch Depth, cm
Active peat extraction site	Site 1 (Lambārtes Mire) Site 2 (Ušuru Mire)	X: 518826; Y: 262233 X: 661175; Y: 324116	143 145	90 123
Abandoned peat extraction site not covered with vegetation (bare peat)	Site 3 (Cenas Mire) Site 4 (Medema Mire)	X: 498792; Y: 297866 X: 506624; Y: 300175	196 188	65 45
Abandoned peat extraction site with shrub and herbaceous plant vegetation	Site 5 (Cenas Mire) Site 6 (Cepļa Mire)	X: 498615; Y: 298016 X: 649492; Y: 344598	130 204	69 59
Scots pine forest	Site 7 (Cepļa Mire) Site 8 (MPS Mežole)	X: 649724; Y: 344213 X: 620173; Y: 349117	214 217	55 33
Silver birch forest	Site 9 (Pleces Mire) Site 10 (MPS Mežole)	X: 348265; Y: 289795 X: 624262; Y: 354836	260 264	58 52

2.2. GHG Measurements

GHG flux measurements were done once per month during 2021 except the frost period of winter (from March to December randomly between 9:30 and 16:00) in 3 replicates in each ditch (distance between replicates was 10–25 m). To measure GHG fluxes, we used a closed-type GHG flux measurement chamber, which—perpendicular to the longitudinal axis of the drainage ditch—covers its entire surface, providing GHG flux measurements from a full cross-sectional area including ditch bed or water surface and slopes (ditch sides). Cross-sectional area of ditches (plane at soil surface) ranged from 0.65 m^2 to 1.32 m^2. The cover frame was made from metal constructions to which a durable, opaque plastic film was attached; the outer side was white and the inner side was black to reflect sunlight and minimize internal temperature fluctuations in the chamber. The metal construction covered the surface of the drainage ditch, while the plastic film was pressed to the ditch profile using a stainless steel chain placed along the perimeter of the gas exchange chamber. The chamber can be used for GHG flux measurements from drainage ditches with different depths, widths, profiles, and water levels, as its length and height can be changed, ensuring the possibility of performing measurements in different environmental conditions. During the measurements, selected width (50 cm constantly), height, and length of the chamber were fixed. Inside the GHG flux measurement chamber, there was a small ventilator installed to ensure that air inside the chamber was continuously mixed. A portable Fourier Transform Infrared (FTIR) spectroscopy (Gasmet DX4040 gas analyzer [35]) was used to measure GHG fluxes. GHG flux measurements—changes in the average content of CO_2, CH_4, and N_2O in atmosphere enclosed in the chamber within 2 min time intervals for 30 min period (respectively, every measurement period was characterized by 15 individual measurements per chamber)—were recorded using software "Calcmet Lite v2.0" [35].

2.3. Measurements of Environmental Variables

At each GHG measurement event, environmental variables were measured. These variables included GW level and the soil and air temperatures, which were measured using Comet Data Logger sensors (Comet System s.r.o., Roznov pod Radhostem, Czech Republic) [36], and the atmospheric pressure, which was measured using Gasmet DX4040 (Gasmet Technologies Oy, Vantaa, Finland) [35]. The water level in drainage ditches was also measured (zero means that the ditch was dry). Three GW wells were sunk in each research site next to the ditch: Positive values mean that the water level was below the soil surface, negative that the water level is was above the soil surface (that is, the area was flooded). Cloudiness, windiness and atypical environmental conditions were fixed.

In addition, GW was sampled at each GHG measurement event and the samples were transported to the LVS EN ISO 17025:2018 accredited laboratory at the Latvian State Forest Research Institute "Silava" and prepared for analysis. The following general chemistry parameters were determined: pH according to LVS ISO 10523:2012; electrical conductivity (EC) according to LVS EN 27888:1993; total nitrogen (TN) and dissolved organic carbon (DOC) concentrations were determined using a FORMACSHT TOC/TN Analyser (ND25 nitrogen detector) according to LVS EN 12260:2004 and to LVS EN 1484:2000; and potassium (K), calcium (Ca), and magnesium (Mg) concentrations in water were determined using the flame atomic absorption spectroscopy (Thermo Fisher Scientific iCE3500, Thermo Fisher Scientific (Asheville) LLC, USA, Serial No: AA05191115) according to LVS EN ISO 7980:2000 and LVS ISO 9964-3:2000. Water samples from ditches were not collected due to the ditches being empty for most of the year.

2.4. Statistical Analysis

All statistical analyses were carried out using the R [37]. A Kruskal–Wallis rank sum test and pairwise comparisons using the Wilcoxon rank sum exact test were used to evaluate possible differences in the mean values of GHG emissions and environmental variables, including GW chemistry between different groups (for instance, groups of current peatland uses), with a significance level of 0.05. Correlations between GHG emissions and different environmental variables were tested with Spearman's ρ (R package "corrplot" [38]), using a significance level of 0.05 (the function rcorr() from R package "Hmisc" was used to compute the significance levels for Spearman correlations [39]).

Environmental variables such as temperature, water level in ditches, GW level and general chemistry (X) were used to explain the variance of instantaneous GHG emissions from drainage ditches (Y) in partial least squares (PLS) regression—a useful multivariate method for dealing with variables that are linearly related to each other, as this method is robust against intercorrelations among X-variables. R package "mdatools" [40] was used to compute the PLS regression. In PLS, X variables are ranked according to their relevance in explaining Y, commonly expressed as variables important for projection (VIP values). Only X variables with VIP values exceeding 0.5 were used in PLS regression, and VIP values exceeding 1.0 are considered as important X variables [41–43].

3. Results

3.1. Variation of Instantaneous GHG Emissions among Different Type of Peatlands and Across Seasons

CO_2 emissions from drainage ditches (Figure 2) tended to be higher in peatland forests where episodic, exceptionally high values of instantaneous CO_2 emissions were observed during the summer and spring seasons (ranging from 3.1 mg CO_2-C m^{-2} h^{-1} in the frost-free period of the winter season to 727.1 mg CO_2-C m^{-2} h^{-1} in summer) compared to peat extraction sites (ranging from −4.6 mg CO_2-C m^{-2} h^{-1} in autumn to 83.8 mg CO_2-C m^{-2} h^{-1}, also in autumn). In peatland forests, the mean value of instantaneous CO_2 emissions was 136.6 ± 28.7 mg CO_2-C m^{-2} h^{-1} (median value 61.0 mg CO_2-C m^{-2} h^{-1}), while in peat extraction sites the mean value was 14.7 ± 2.4 mg CO_2-C m^{-2} h^{-1} (median value 10.1 mg CO_2-C m^{-2} h^{-1}). Mean values of CO_2 emissions from each individual ditch revealed that all studied drainage ditches acted as sources of CO_2 emissions to the atmosphere. Furthermore, the CO_2 emissions in summer were significantly higher than the CO_2 emissions recorded in autumn, winter, and spring (p = 0.008, p = 0.002, p = 0.008, respectively).

Instantaneous CH_4 emissions from drainage ditches (Figure 2) ranged from −2.2 mg CH_4-C m^{-2} h^{-1} in abandoned peat extraction sites with shrub and herbaceous plant vegetation in summer to 12.6 mg CH_4-C m^{-2} h^{-1} in abandoned peat extraction sites with bare peat, also in summer. The mean value of CH_4 emissions was 0.085 ± 0.034 mg CH_4-C m^{-2} h^{-1} (median value 0.024 mg CH_4-C m^{-2} h^{-1}) in peatland forests, while in peat extraction sites the mean value was 1.07 ± 0.45 mg CH_4-C m^{-2} h^{-1} (median value 0.035 mg CH_4-C m^{-2} h^{-1}). Mean values of CH_4 emissions from each individual ditch revealed that most of the studied drainage ditches acted as sources of CH_4 emissions to the atmosphere, except for three ditches where a removal of CH_4 was observed (minimum mean CH_4 emission value was −0.35 mg CH_4-C m^{-2} h^{-1} in abandoned peat extraction site with shrub and herbaceous plant vegetation). Significant seasonality impact on CH_4 emissions was not observed.

The highest instantaneous N_2O emissions (Figure 2) were found in silver birch forests (ranging from −0.004 mg N_2O-N m^{-2} h^{-1} in summer to 0.076 mg N_2O-N m^{-2} h^{-1} in spring) compared to peat extraction sites (ranging from −0.107 mg N_2O-N m^{-2} h^{-1} in summer to 0.065 mg N_2O-N m^{-2} h^{-1} in summer) and Scots pine forests (ranging from −0.009 mg N_2O-N m^{-2} h^{-1} in summer to 0.010 mg N_2O-N m^{-2} h^{-1} in spring). Mean value of N_2O emissions was 0.009 ± 0.003 mg N_2O-N m^{-2} h^{-1} (median value 0.001 mg N_2O-N m^{-2} h^{-1}) in peatland forests, while in peat extraction sites the mean value was −0.003 ± 0.004 mg N_2O-N m^{-2} h^{-1} (median value −0.001 mg N_2O-N m^{-2} h^{-1}). Mean values of N_2O emissions from each individual ditch revealed that half (50%) of the studied drainage ditches acted as sources of N_2O emissions to the atmosphere (the maximum mean N_2O emission value was 0.033 mg N_2O-N m^{-2} h^{-1} in the silver birch forest), but the other half of the studied drainage ditches acted as sinks of N_2O emissions (the minimum mean N_2O emission value was −0.013 mg N_2O-N m^{-2} h^{-1} in abandoned peat extraction site with bare peat). As with CH_4 emissions, significant differences in the N_2O emissions between seasons were not found (p > 0.75).

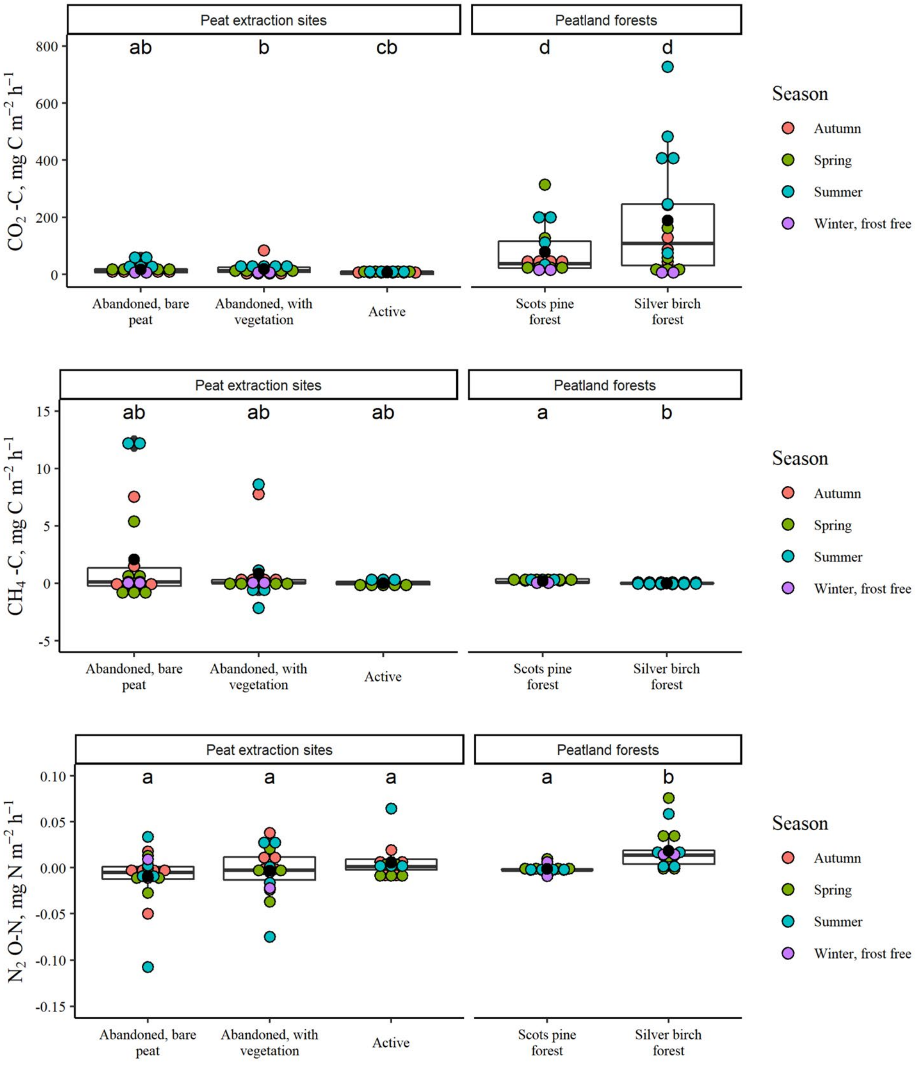

Figure 2. Variation of instantaneous GHG emissions from drainage ditches in peatlands by types of current land use. In the boxplots, the median is shown by the bold line, the mean by the black dot. The box corresponds to the lower and upper quartiles, and the whiskers show the minimal and maximal values (within 150% of the interquartile range from the median), while dots outside the box represent outliers of the datasets. Colored dots represent different seasons, and different lowercase letters show statistically significant differences ($p < 0.05$) in mean values between groups of current peatland uses. Figure was prepared with the R package "ggplot2" [44].

The comparison of the contributions of different GHG emission in terms of warming potential is given in Figure 3, where CH_4 and N_2O emissions have been recalculated to CO_2 equivalents (CH_4 and N_2O is 25 and 298 times as potent as CO_2, respectively). In all ditches, except ditches in abandoned peat extraction sites with bare peat, the dominant GHG in terms of warming potential was CO_2. The contribution of CH_4 emissions from drainage ditches in silver birch forests and active peat extraction sites (−0.43 and −0.63 mg CO_2-eq. m^{-2} h^{-1}, respectively), as well as the contribution of N_2O emissions from drainage ditches in Scots pine forests and abandoned peat extraction sites with shrub and herbaceous plant vegetation (−0.58 and −1.39 mg CO_2-eq. m^{-2} h^{-1}, respectively) was negligible.

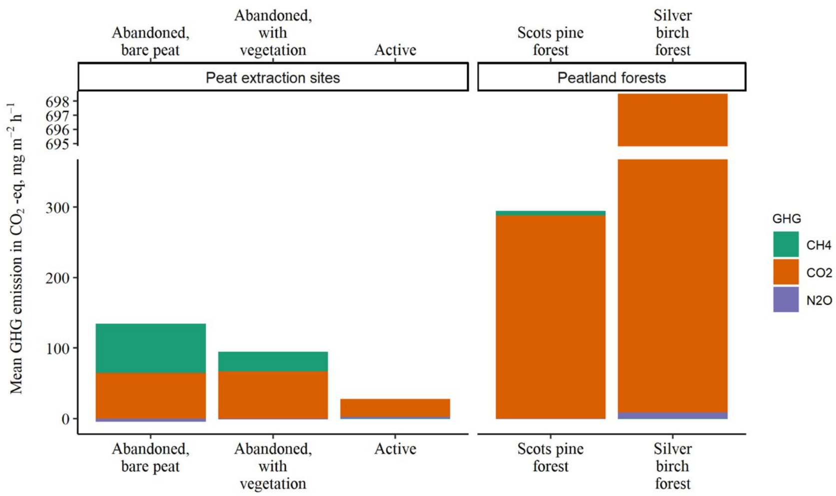

Figure 3. Contribution to greenhouse warming of different GHG emissions from drainage ditches, given in CO_2 equivalents. Figure was prepared with the R package "ggplot2" [44] and "ggbreak" [45].

3.2. Evaluation of Affecting Factors

The instantaneous CO_2 emissions from drainage ditches both in peatland forests and peat extraction sites were positively correlated with water temperature (ρ = 0.68 and ρ = 0.37, respectively) and negatively with water level in ditches (ρ = −0.43 and ρ = −0.30, respectively) (Figures 4 and 5). Furthermore, in peatland forests, the average CO_2 emissions from dry ditches were significantly higher than from water-filled ditches ($p < 0.001$) (Figure 6). In peat extraction sites, there was no detectable difference in the average CO_2 emissions between water-filled and dry ditches. In addition, in peatland forests, CO_2 emissions were positively correlated with the GW level (ρ = 0.76), K and TN concentrations in water (ρ = 0.66 and ρ = 0.48, respectively), and air temperature (ρ = 0.54) (Figures 4 and 5). A PLS model ($R^2 = 0.64$, $Q^2 = 0.41$) revealed that the variation in instantaneous CO_2 emissions from drainage ditches in peatland forests was generally explained by the GW level (VIP = 1.5), water temperature (VIP = 1.3), K concentrations in water (VIP = 1.2), and air temperature (VIP = 1.0), while in peat extraction sites a PLS model was weak ($R^2 = 0.26$, $Q^2 < 0.10$).

Instantaneous CH_4 emissions from drainage ditches in peatland forests were negatively correlated with the GW level (ρ = −0.57), pH (ρ = −0.57), and K and TN concentration in water (ρ = −0.60 and ρ = −0.46, respectively), and positively with the water level in ditches (ρ = 0.48). In peat extraction sites, CH_4 emissions from drainage ditches were positively correlated with water temperature (ρ = 0.36) (Figures 4 and 5). Although higher average CH_4 emissions were recorded from water-filled ditches compared to dry ditches, there was no significant difference in CH_4 emissions between water-filled and dry ditches (Figure 6). A PLS model (R^2 = 0.60, Q^2 = 0.32) revealed that the variation in instantaneous CH_4 emissions from drainage ditches in peatland forests was generally explained by the GW level (VIP = 1.5), water level in ditches (VIP = 1.1), and K concentrations in water (VIP = 1.1), while in peat extraction sites a PLS model was very weak (R^2 = 0.21, Q^2 < 0.10).

Instantaneous N_2O emissions from drainage ditches in peatland forests were positively correlated with the GW level, and K and TN concentration in water (ρ = 0.63, ρ = 0.63 and ρ = 0.48, respectively) (Figures 4 and 5). In peat extraction sites, there was no detectable difference in the average N_2O emissions between water-filled and dry ditches, while in peatland forests the average N_2O emissions from dry ditches were significantly higher than from water-filled ditches (p = 0.021), similar to the case of CO_2 emissions (Figure 6). PLS models explaining variation in instantaneous N_2O emissions from drainage ditches were weak both in peatland forests (R^2 = 0.48, Q^2 = 0.14) and peat extraction sites (R^2 < 0.10, Q^2 < 0.10).

In addition, cross-correlations were found between GHG emissions. While CO_2 emissions from drainage ditches tended to correlate positively with N_2O emissions, both CO_2 and N_2O emissions simultaneously tended to correlate negatively with CH_4 emissions (Figure 4). Variations and mean values of water level in ditches, GW level, and parameters of GW chemistry in research sites are summarized in Table 2.

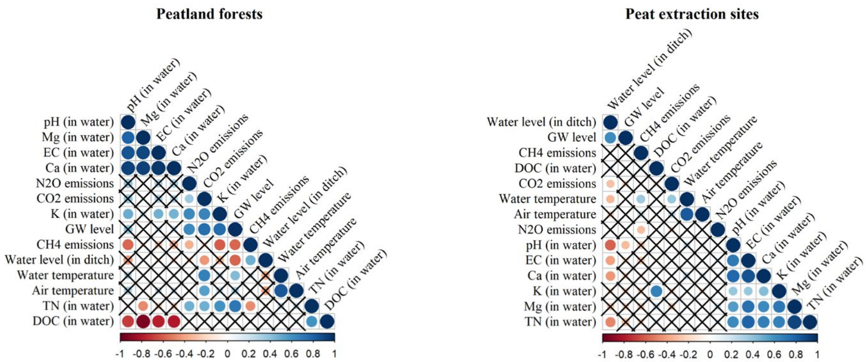

Figure 4. Spearman's correlations between instantaneous GHG emissions from drainage ditches and environmental variables (temperatures, water level in ditches, GW level, and general chemistry). Positive correlations are displayed in blue and negative correlations in red. Color intensity and the size of the circle are proportional to the correlation coefficients. Below the correlogram, the legend color shows the correlation coefficients and the corresponding colors. Correlations with p > 0.05 are considered insignificant (crosses are added). Figure was prepared with the R packages "corrplot" [38] and "Hmisc" [39].

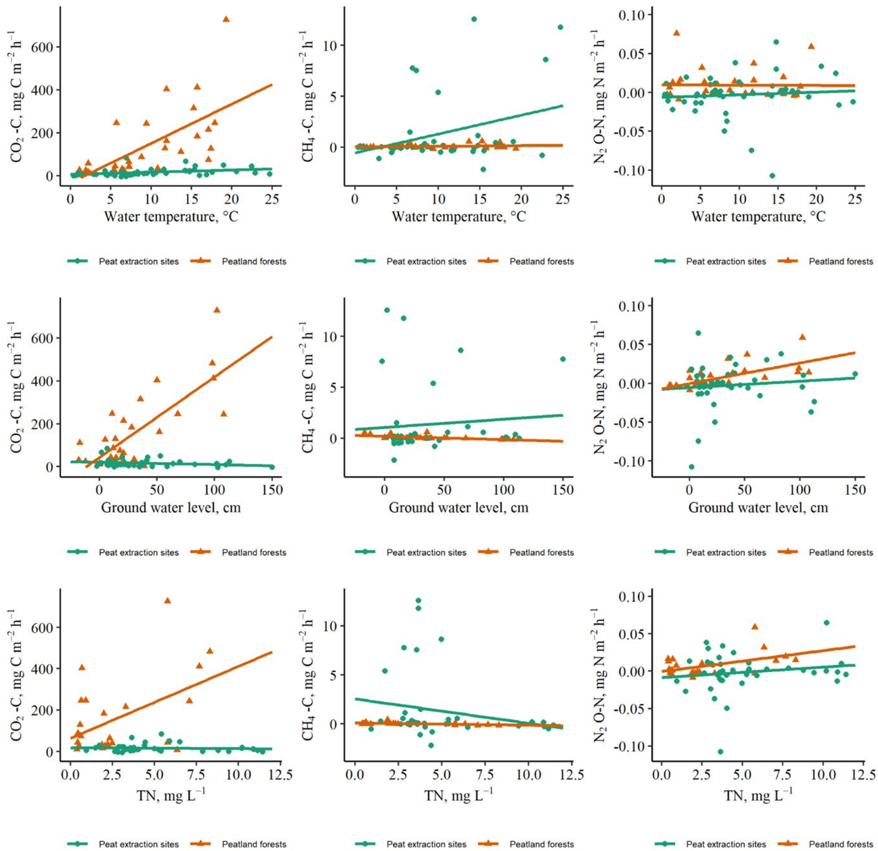

Figure 5. Relationships between instantaneous CO_2, CH_4, and N_2O emissions from drainage ditches and water temperature, groundwater level below soil surface, and total nitrogen concentration in groundwater. Figure was prepared with the R package "ggplot2" [44].

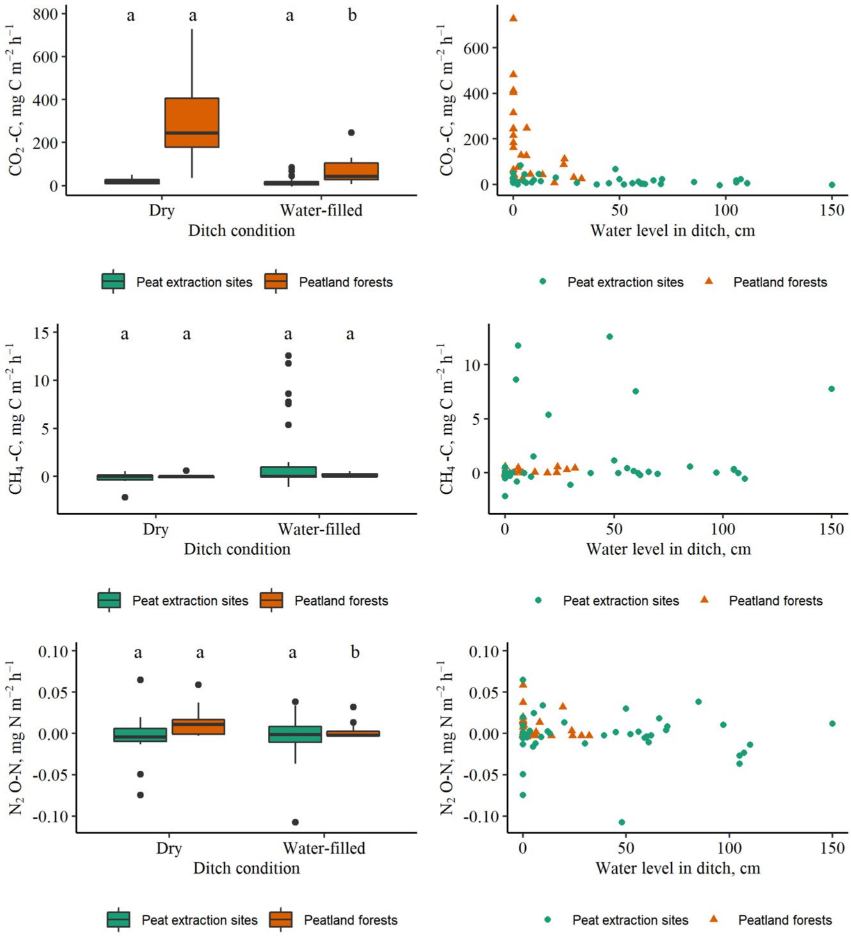

Figure 6. Variation of instantaneous GHG emissions from drainage ditches depending on water level in ditches. In the boxplots, the median is shown by the bold line. The box corresponds to the lower and upper quartiles, whiskers show the minimal and maximal values (within 150% of the interquartile range from the median), and dots outside the box represent outliers of the datasets. In the boxplots, different lowercase letters show statistically significant differences ($p < 0.05$) in the mean values between the dry and water-filled ditches within the groups of current land use of peatland (peat extraction sites and peatland forests). Figure was prepared with the R package "ggplot2" [44].

Table 2. Variations and mean values of water level in ditches, GW level, and parameters of GW general chemistry during the study period in research sites. Different lowercase letters show statistically significant differences ($p < 0.05$) in mean values between groups of current land use of peatland.

Parameter, Unit	Value	Peat Extraction Sites			Peatland Forests	
		Abandoned, Bare Peat	Abandoned, with Vegetation	Active	Scots Pine Forest	Silver Birch Forest
Water level in ditch, cm	mean ± S.E. range	37.6 ± 8.8 [a] 0–110	44.6 ± 12.9 [ab] 0–150	26.6 ± 8.1 [ab] 0–70	8.7 ± 3.4 [ab] 0–32	4.8 ± 2.0 [b] 0–24
GW level, cm	mean ± S.E. range	18.0 ± 4.1 [ab] −2–53	62.1 ± 12.0 [a] 7–150	24.1 ± 4.4 [ab] 8–59	10.6 ± 4.7 [b] −18–36	45.4 ± 9.6 [ab] 0–108
pH	mean ± S.E. range	5.2 ± 0.2 [a] 3.8–7.0	5.2 ± 0.3 [a] 3.9–6.7	6.3 ± 0.2 [b] 5.2–7.0	5.2 ± 0.3 [a] 3.8–6.4	7.3 ± 0.1 [c] 6.4–7.9
EC, $\mu S\ cm^{-1}$	mean ± S.E. range	48.4 ± 5.5 [ad] 32.7–117.2	64.1 ± 9.5 [bcd] 38.9–160.6	110.9 ± 16.5 [cd] 34.4–225	64.1 ± 12.8 [d] 36.5–163.25	292.9 ± 24.6 [e] 142.7–450.5
TN, $mg\ L^{-1}$	mean ± S.E. range	3.57 ± 0.33 [abde] 0.94–6.52	3.84 ± 0.27 [be] 2.78–5.42	8.27 ± 0.86 [c] 2.65–11.43	2.57 ± 0.38 [de] 1.86–5.77	2.88 ± 0.87 [e] 0.39–8.29
DOC, $mg\ L^{-1}$	mean ± S.E. range	99.1 ± 6.7 [a] 49.1–140.8	101.9 ± 5.8 [a] 81.4–142.0	125.8 ± 22.5 [a] 64.0–291.7	109.3 ± 6.7 [a] 89.7–149.7	41.2 ± 10.3 [b] 9.2–101.0
K, $mg\ L^{-1}$	mean ± S.E. range	0.94 ± 0.08 [ae] 0.47–1.50	0.65 ± 0.07 [be] 0.23–0.95	2.19 ± 0.28 [c] 0.76–3.96	0.42 ± 0.04 [d] 0.23–0.75	0.91 ± 0.11 [e] 0.38–1.76
Ca, $mg\ L^{-1}$	mean ± S.E. range	10.6 ± 1.7 [abd] 1.78–28.19	13.4 ± 2.3 [bcd] 5.42–31.57	19.5 ± 2.8 [cd] 4.76–39.93	17.0 ± 3.4 [d] 4.14–41.87	62.5 ± 4.1 [e] 34.64–79.93
Mg, $mg\ L^{-1}$	mean ± S.E. range	0.63 ± 0.03 [a] 0.49–0.85	1.19 ± 0.12 [b] 0.83–1.93	4.67 ± 0.52 [c] 0.73–6.66	0.74 ± 0.13 [a] 0.34–1.62	12.82 ± 1.51 [e] 4.82–18.32

4. Discussion

4.1. CO_2 Emissions

In most of the studied drainage ditches, CO_2 was the dominant GHG in terms of greenhouse warming potential. The highest instantaneous CO_2 emissions from drainage ditches were found in peatland forests, especially during the summer season (ranging up to 727.1 mg CO_2-C $m^{-2}\ h^{-1}$) when drainage ditches were dry and GW level was at least 50 cm below soil surface. In peat extraction sites, instantaneous CO_2 emissions ranged from −4.6 to 83.8 mg CO_2-C $m^{-2}\ h^{-1}$. Although in some cases negative CO_2 emissions (CO_2 removals) were found (for instance, in peat extraction sites during the spring and autumn seasons), the mean values of CO_2 emissions revealed that all studied drainage ditches acted as sources of CO_2 emissions to the atmosphere, indicating that CO_2 production exceeded CO_2 uptake during photosynthesis by plants [16].

Episodic, exceptionally high instantaneous CO_2 emissions from drainage ditches recorded in peatland forests in summer and spring significantly increase the mean value of CO_2 emissions, which results in significant differences between the mean and median values of CO_2 emissions. Thus, extrapolation and inclusion of these episodic, exceptionally high instantaneous CO_2 emissions from drainage ditches in calculations of annual CO_2 emissions should be done with caution to avoid overestimating annual CO_2 emissions. As CO_2 emissions from drainage ditches correlate significantly with several environmental variables, e.g., temperature, GW level, and the presence of surface water, the best approach for calculation of the annual CO_2 emissions is very likely to use multivariate equations. Such an approach, however, requires a wide monitoring (activity) data set. Episodic increases in CO_2 emissions from drainage ditches when the ditches were dry can be explained by both increased mineralization of fresh organic matter (for instance, tree litter in peatland forests) in oxic conditions and by the intensification of some of the pathways by which CO_2 was transported into the atmosphere. However, a longer monitoring period of GHG emissions from drainage ditches (at least two years period) and more frequent campaigns would increase knowledge of the contribution of episodic, exceptionally high instantaneous fluxes to the annual emissions, and would improve identification and characterization of the main affecting factors determining GHG emissions.

In general, reported CO_2 emissions from drained ditches vary widely. For instance, Peacock et al. (2021) revealed no significant difference in the mean CO_2 emissions between drainage ditches in catchments with mineral and peat soils in boreal and hemiboreal regions and reported the mean CO_2 emissions of 6016 (range −720 to 32,470) mg CO_2 m^{-2} d^{-1} from drainage ditches in forests in southern Sweden [5]. Hyvönen et al. (2013) reported that the daily CO_2 emission from drainage ditches in a boreal cutaway peatland cultivated with reed canary grass in eastern Finland ranged from −0.4 mg m^{-1} h^{-1} to 468.5 mg m^{-1} h^{-1} [17]. Sundh et al. (2000) reported average CO_2 emissions from drainage ditches in peat-mining areas in Sweden of −112–161 mg CO_2 m^{-2} h^{-1} [18], while Schrier-Uijl et al. (2011) reported that CO_2 emission from the drainage ditches in peat areas in the Netherlands ranged from 69.6 mg m^{-2} h^{-1} to 199.0 mg m^{-2} h^{-1} [16].

Nevertheless, several studies have concluded that the ditches do not contribute significantly to the total site CO_2 emissions (e.g., [17,18]). Our estimates of instantaneous CO_2 emissions from drainage ditches in peat extraction sites did not exceed the ranges reported previously for drained peat soils in peat extraction sites in Latvia (e.g., [46]) and Estonia (e.g., [47]). By contrast, CO_2 emissions from drainage ditches in peatland forests in some months even exceed maximum monthly average total CO_2 emissions from soils in nutrient-rich organic forest soils in Latvia (15.81 t C ha^{-1} yr^{-1}), as recently found by Butlers et al. (2022) [48]. This is explained by the impact of several episodic, exceptionally high records of instantaneous CO_2 emissions in peatland forests in summer and spring, as discussed above.

Research results regarding relationships between CO_2 emissions from ditches and environmental parameters, ditch parameters, presence of vegetation, water chemistry, and other parameters are not unambiguous, but mostly no correlations are reported (e.g., [16–18]). Nevertheless, Schrier-Uijl et al. (2011) found that a higher trophic status correlates positively with CO_2 emissions, while the depth of the water and the pH correlate inversely with CO_2 emissions [16]. Similarly, we found positive correlations between CO_2 emissions from drainage ditches and K and TN concentrations in water, and negative correlations with water levels in the ditches. In addition, we revealed positive correlations between CO_2 emissions from drainage ditches and the GW level (cm below soil surface) and temperatures (water and air). Although CO_2 is highly soluble in water and can be leached [17], the dependence of CO_2 emissions on temperature confirms the existence of biological (microbial) processes that regulate CO_2 emissions [16].

4.2. CH_4 Emissions

CH_4 emissions from the studied drainage ditches ranged from −2.2 mg to 12.6 mg CH_4-C m^{-2} h^{-1}. The highest recorded instantaneous CH_4 emissions can be characterized as episodic, exceptionally high emissions most likely caused by bubble ebullition [20,26,27]. Most of the studied drainage ditches acted as sources of CH_4 emissions to the atmosphere, except for a few ditches where small CH_4 removals were observed. However, our estimates are in the range of the CH_4 emissions from drainage ditches reported by other studies. For instance, Peacock et al. (2021) reported a CH_4 emission range from 0.1 to 386 g CH_4 m^{-2} y^{-1} with a mean of 64.6 ± 11.1 g CH_4 m^{-2} y^{-1} based on a literature synthesis covering both boreal, temperate, and tropical climate zones [7]. In Sweden, the mean CH_4 emissions of 33.9 (range −1.3 to 1390) mg CH_4 m^{-2} d^{-1} were reported from drainage ditches in forests [5], while in peat-mining areas CH_4 emissions from drainage ditches reached 93 mg CH_4 m^{-2} h^{-1} with a mean rate of 15.1 ± 23.9 mg CH_4 m^{-2} h^{-1} [18]. In Finland, daily CH_4 emissions from drainage ditches in a boreal cutaway peatland cultivated with reed canary grass ranged from –1.87 mg m^{-2} d^{-1} to 99.32 mg m^{-2} d^{-1} [17], while CH_4 emissions from drainage ditch bottoms and ditch sides in Lakkasuo mire (central Finland) ranged from 0 to 595 and from 0 to 78 mg m^{-2} d^{-1}, respectively; furthermore, the highest emissions were measured from the ditch bottoms covered by water [2]. Hyvönen et al. (2013) also highlighted that waterlogged ditches showed the highest CH_4 emissions, which the authors explained by their having anaerobic conditions that favor CH_4 production but limit CH_4 oxidation [17]. Moore and Roulet (1993), Liblik et al. (1997) in Canada [49,50], and van den Pol-van Dasselaar (1998) in the Netherlands [51] found strong relationships between the average seasonal CH_4 emissions and GW level. Our results support this relationship: CH_4 emissions from water-filled ditches were higher than from dry ditches (although the difference was not significant). As well, we also found a positive correlation between CH_4 emissions from drainage ditches and water levels in ditches, and a negative correlation between CH_4 emissions and the GW level.

Several studies have highlighted that CH_4 emissions from ditches tend to increase with temperature and that higher CH_4 emissions were found from more eutrophic ditches (e.g., [7,16,51]. The impact of temperature is related to the decreased activity of methanogens and other bacteria implied by methanogenic fermentation at low temperatures [28]. Our results also showed a positive correlation between CH_4 emissions from drainage ditches and water temperature; in contrast, we found a negative correlation between CH_4 emissions and TN and K concentrations in water, which indirectly indicates the trophic status of the water. Although the activity of methanogens producing CH_4 is optimum around neutrality or under slightly alkaline conditions, methanogens can partly adapt to acidic environments [28,52]. A negative correlation between CH_4 emissions from drainage ditches and water pH was also found. The mean GW pH over the study period did not drop below 5.2 at our research sites, indicating that the environment in the research sites was not extremely acidic, which could have limited CH_4 production.

We found no significant differences in CH_4 emissions from ditches of peat extraction sites and those of peatland forests. A similar observation was made by Peacock et al. (2021) [7]. An earlier study from Latvia [46] and Estonia [47] demonstrated that CH_4 emissions from drained peat soils (ranging from −32.12 to 170.44 μg CH_4-C m^{-2} h^{-1} in Latvia and from −82 to 12,037 μg CH_4-C m^{-2} h^{-1} in Estonia) were significantly lower in Latvia and similar in Estonia to those from the drainage ditches recorded within this study. The finding confirms that CH_4 emissions from the ditches can contribute significantly to the total site CH_4 emissions, including emissions from the peat soils and drainage ditches as reported by, for instance, Peacock et al. (2021) [5], Sundh et al. (2000) [18], and Roulet and Moore (2011) [21].

IPCC (2014) provided CH_4 emission factors for drainage ditches in forest land with drained organic soils and peat extraction sites of 217 and 542 kg CH_4 ha^{-1} yr^{-1}, respectively, in boreal and temperate climate zones [6]. Our mean CH_4 emission factor was 10.3 kg CH_4 ha^{-1} yr^{-1} for drained peatland forests and 122.5 kg CH_4 ha^{-1} yr^{-1} for peat extraction sites with the highest annual CH_4 emissions in abandoned peat extraction sites with bare peat (244.3 kg CH_4 ha^{-1} yr^{-1}). Although our estimates demonstrated that annual CH_4 emissions (emission factors) for drainage ditches in hemiboreal peatlands are notably smaller than those provided by the IPCC guidelines [6], the calculated CH_4 emission factor for peat extraction sites lay in the uncertainty range of the IPCC default emissions factor (102–981 kg CH_4 ha^{-1} yr^{-1}). Moreover, estimates within this study demonstrated a significantly narrower range of variation of annual CH_4 emissions from drainage ditches than that provided by the IPCC guidelines. However, our annual CH_4 emissions were calculated as the mean of instantaneous CH_4 emissions expressed in annual units (yr^{-1}) including episodic, exceptionally high instantaneous CH_4 emissions identified in abandoned peat extraction sites. As with CO_2 emissions, the best approach to calculate annual CH_4 emissions would very likely be to use multivariate equations that would avoid potential overestimations.

4.3. N_2O Emissions

N_2O emissions from the drainage ditches in the studied research sites were negligible in terms of greenhouse warming potential and ranged from −0.107 to 0.076 mg

N_2O-N m^{-2} h^{-1}. The highest mean N_2O emissions from ditches were found in silver birch forests (the mean value of 0.033 mg N_2O-N m^{-2} h^{-1}). Negative N_2O emission values can be explained by complete denitrification, resulting in N_2O conversion into inert N_2 under anaerobic conditions [53]. It is supported by the findings in peatland forests, where the average N_2O emissions from dry ditches were significantly higher than from water-filled ditches. As Hyvönen et al. (2013) [17] also found, a significant temporal variation of N_2O emissions from ditches (difference between seasons) was not found, nor was temperature found to be a significant factor affecting N_2O emissions.

An earlier study from Latvia [46] demonstrated that N_2O emissions from drained peat soils were even lower and varied in a narrower range from −0.001 to 0.013 mg N_2O-N m^{-2} h^{-1} compared to emissions from drainage ditches. Findings in Estonia [47], however, revealed that N_2O emissions from peat soils (the average reported emissions of N_2O-N varied between −22.7 and 328.8 µg N_2O-N m^{-2} h^{-1}) were higher than from the drainage ditches recorded in this study.

4.4. Impact of Afforestation of Peat Extraction Areas

There is some evidence of increasing tree cover both in pristine and managed boreal and temperate peatlands due to changes in climate and land use [54,55]. Increased tree cover in peatlands has a strong impact on the peat's physical, chemical, and microbial properties [54,55] and consequently on biogeochemical cycling of elements including GHG fluxes. The results of this study indirectly demonstrated that afforestation of drained peat extraction areas would most likely lead to increased GHG emissions from drainage ditches, although only CO_2 and N_2O emissions were observed to be higher in peatland forests (especially in more fertile silver birch stands) than in peat extraction sites. Most probably, increased CO_2 and N_2O emissions from drainage ditches in peatland forests compared to peat extraction sites can be a result of mineralization of fresh tree litter especially in oxic conditions when ditches were dry. Furthermore, decomposition of litter has been faster in the deciduous stands than in the coniferous stands (e.g., [56–60]) and this may generally be explained by higher lignin content in coniferous litter (e.g., [56,57]). The higher CO_2 and N_2O emissions from drainage ditches in silver birch forests compared to the Scots pine forests observed in this study support this interpretation.

Nevertheless, the potential increase in GHG emissions from drainage ditches after afforestation of former peat extraction areas could be compensated with CO_2 sequestration in tree biomass and other C pools. Recent findings also demonstrated that soils in drained and afforested peatlands can be a net sink for C (considering C input through tree litter and forest floor vegetation as well), since the amount of C entering the soil can substantially exceed the C released due to the heterotrophic decomposition of soil organic matter [59,60].

5. Conclusions

In terms of warming potential, the contribution of CO_2 emissions to the total budget of GHG emission from ditches in drained peatlands can be higher than the CH_4 contribution. For this reason, both GHGs must be considered (included) in calculations of a total landscape-level GHG budget. Average instantaneous CO_2 and N_2O emissions from drainage ditches in peatland forests were significantly higher than those from ditches in peat extraction sites, while there was no difference in the average CH_4 emissions from ditches between peatland forests and peat extraction sites.

Emissions from ditches of all GHGs increased with increasing temperature. In addition, CO_2 and N_2O emissions from drainage ditches increased with a fall in the GW level. They were also negatively correlated with water levels in ditches, but positively correlated with K and TN concentrations in water. By contrast, CH_4 emissions from drainage ditches increased with increased GW level and water levels in ditches, but were negatively correlated with K and TN concentrations in water and water pH.

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Appendix A

Table A1. Characterization of the forest stands (afforested peatland after peat extraction).

Current Land Use/Type of Vegetation	Research Site	Tree Diameter at Breast Height (Mean ± S.E.), cm	Tree Height (Mean ± S.E.), m
Scots pine forest	Site 7 (Cepļa Mire)	7.9 ± 0.4	8.6 ± 0.5
	Site 8 (MPS Mežole)	21.3 ± 0.9	18.8 ± 1.3
Silver birch forest	Site 9 (Pleces Mire)	14.2 ± 0.8	13.7 ± 1.4
	Site 10 (MPS Mežole)	15.3 ± 0.3	16.8 ± 0.7

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Carbon Budget of Undrained and Drained Nutrient-Rich Organic Forest Soil

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Abstract: The impact of the moisture regime on the carbon budget of organic soils with different nutrient statuses has not been fully studied in hemiboreal forests thus far. This study evaluated soil carbon (C) stock changes in forests with drained and undrained nutrient-rich organic soils by estimating C loss through respiration and C input through the litter. The study sites included forest stands dominated by Norway spruce (*Picea abies*), silver birch (*Betula pendula*), black alder (*Alnus glutinosa*), and clear-cuts. Soil respiration was measured using the chamber method, and to estimate the soil C input by litter—the biomass and the C content of the foliar litter, ground vegetation, and fine-root production were measured. The soil in forest stands acted as a C sink. The carbon dioxide (CO_2) removal rates of 0.4 ± 0.4 t C ha^{-1} year^{-1} and 0.1 ± 0.4 t C ha^{-1} year^{-1} were estimated for undrained and drained soil in forest stands, respectively. The soil in the clear-cuts acted as a CO_2 source, and the annual emissions ranged from 0.4 ± 0.4 t C ha^{-1} year^{-1} in undrained to 0.9 ± 0.7 t C ha^{-1} year^{-1} in drained conditions. The reason for the soil in clear-cuts being a C source was increased C loss by respiration and reduced soil C input by litter. Furthermore, the mean soil C input by ground vegetation biomass in the clear-cuts was considerably higher than in the forest stands, which did not compensate for the increase in soil respiration and the absence of C input by foliar litter and the fine roots of trees. The results of the study on annual soil C stock changes can be used as an emission factor in national greenhouse gas inventories of forest land in the hemiboreal zone.

Keywords: nutrient-rich organic soil; drainage; soil respiration; litterfall; ground vegetation; fine roots; soil carbon stock changes

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1. Introduction

In accordance with the Paris Agreement, the European Union, including Latvia, has committed to achieving climate neutrality by 2050. These policy targets promote seeking forest management practices that contribute to C sequestration [1–4], reflected in the topicality of related studies, including the estimation of ecosystem greenhouse gas (GHG) balance. Reducing the GHG emissions from the main national sources, such as the transport, energy, and agriculture sectors, which currently account for around 88% of Latvia's total GHG emissions, will not be sufficient to achieve the climate neutrality target set by Paris Agreement. To compensate for the irreducible GHG emissions in these sectors, the land use, land-use change, and forestry (LULUCF) sector must ensure the equivalent sequestration of carbon dioxide (CO_2). Boreal forests are often identified as ecosystems with carbon (C) sequestration potential [5]. Therefore, forest land is the only land-use category of the LULUCF sector in which an increased rate of CO_2 sequestration by the implementation of climate change mitigation measures has the potential to offset the country's total GHG emissions.

Efforts to achieve the C sequestration and GHG mitigation potential of the forest ecosystem can be implemented with targeted activities that promote C sequestration in biomass, soil, and harvested wood products, as well as by replacing fossil fuels with biomass. The role of forest management in tree biomass C sequestration is well understood and modeled, but understanding the process of C sequestration and assessing changes in the C stock of the soil remain limited [6]. Previous studies show that climate change mitigation measures targeted at organic soil management are often the most effective, but the climate change mitigation potential of organic soils is not being used fully [7,8]; this is largely related to the lack of knowledge. The most frequently identified climate change mitigation measures related to the management of organic forest soils are the afforestation of organic soils and the restoration of the natural moisture regime [9]. However, there is a lack of scientific evidence that the restoration of the natural moisture regime of organic soils promotes reductions in GHG emissions and an increase in the C sink of the boreal forest ecosystem in the current climate conditions. GHG emissions from undrained forest soil are not within the scope of the national GHG inventory (Inventory) reports, hindering interest in such studies. However, quantitative awareness of such emissions is crucial for the comparison of carbon stock change (CSC) of drained organic soil relative to undrained soil to fully understand the climate impact of drainage and to enable possibilities of implementing the potentially most effective climate change mitigation measures in forest land management. The drainage of organic soils is often considered a climate-harmful management practice, although knowledge of annual soil GHG emissions is highly uncertain [8]. Currently, there is a lack of common understanding of the impact of soil drainage on forest ecosystem GHG emissions and the C balance. Some studies indicate that the drainage of nutrient-poor organic soils in boreal forests has a significant impact on ecosystem CO_2 sequestration [10], while the drainage of nutrient-rich organic soils may turn forest ecosystems into GHG emission sources when soil C and nitrogen (N) loss is not compensated by increased forest growth [11].

The most commonly mentioned shortcoming of previous scientific articles on net CO_2 emissions from forests with drained organic soils in boreal and temperate climate regions is a necessity to subtract below- and above-ground biomass respiration from the reported results and incorporate litter production or decomposition rates [8]. Thus, the results reported require further processing or additional data to enable the quantification of annual soil CSC. Another shortcoming is the uneven site spatial coverage of the previous studies. Most of the organic soil CSC estimate results were obtained from drained boreal peatland studies carried out in Finland, while most of the study sites representing a temperate zone are located in the southern part of Sweden [8]. The results of organic soil CSC estimates in the Baltic states representing hemiboreal forests are reported by four articles on drained peatlands [12–15]. Despite the fact that the availability of study results on drained organic forest soils has increased, they are still scarce, considering the variability of the factors affecting CSC in forest ecosystems. No CSC estimates of undrained organic forest soils have been reported in the region, as studies on undrained organic soils are usually carried out in pristine or recently recultivated peatlands.

According to the acknowledgment that there is a lack of studies evaluating the impact of different long-term soil moisture regimes on soil CSC [8] and observations that organic forest soil CO_2 emissions can be comparably higher in forest sites with increased soil fertility [16], room for improvement in the Inventory and capabilities to plan climate change mitigation measures is recognized. The estimated GHG emissions of 1.7 million t CO_2 equivalents (14.4% of the total emissions of Latvia) from drained organic soil in the forest land category in 2020 [17] show the significance of accurate organic soil emission estimates in the national Inventory. The currently applied country-specific emission factor (0.52 t C ha^{-1} $year^{-1}$) for the estimation of CO_2 emissions from drained organic soil in forest lands in Latvia is developed by the C stock inventory method conducted in forests with nutrient-poor to moderately rich (*Callunosa turf. Mel.*, *Vacciniosa turf. Mel*. Additionally, *Myrtillosa turf. Mel.*) soils according to the national forest site type classification [18]. The country-specific emission factor is applied to all organic soils in forests, while according to the national forest inventory, the share of drained (17%) and undrained (4%) forest site types with nutrient-rich organic soils, where potentially higher soil CO_2 emissions may be expected, is 21%. Therefore, the currently used emission factor may introduce accuracy errors in the estimations.

This study aimed to estimate the CSC of drained and undrained nutrient-rich organic soils using empirical data on soil CO_2 emissions and soil C input by:

- Foliar litter (LF);
- Ground vegetation (above- and below-ground biomass of herbs and grasses, GV);
- Fine roots of trees (FR);
- Moss and dwarf shrubs.

This study contributes to the improvement of the national GHG inventory and provides a scientifically valid assessment of potential soil drainage effects on CO_2 emissions to support decision-making on climate change mitigation measures.

2. Materials and Methods

2.1. Study Site Description

The study was carried out in central Latvia (Figure 1) on the forest stands of a hemiboreal zone with undrained (*Dryopterioso-caricosa* and *Filipendulosa*) and drained (*Oxalidosa turf. mel.*) forest site types characterized by nutrient-rich organic soil. For the forest stands to be accepted as study sites, the compliance with drainage status, the average peat layer depth (>30 cm in undrained sites and >20 cm in drained sites), and the characteristic vegetation, as defined in the national forest site type classification (Table 1), were analyzed. One round sample plot (500 m^2) was established in each of the selected study sites. The distance to the nearest drainage ditches from the sample plots was at least 300 m and 100 m in the study sites with undrained and drained soil, respectively.

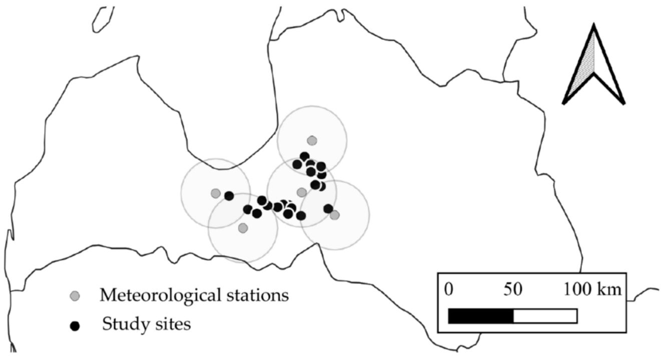

Figure 1. Location of study sites and closest meteorological stations with the indicated radius of 30 km.

Table 1. Dominant ground vegetation in the study sites.

Forest Site Type	Ground Vegetation
Dryopterioso-caricosa	*Thelypteris palustris* Schott, *Carex* (L.), *Iris pseudacorus* (L.) Fuss, *Scirpus* (L.), *Lysimachia vulgaris* (L.), *Cirsium oleraceum* (L.) Scopoli, *Filipendula ulmaria* (L.) Maximowicz, *Angelica sylvestris* (L.)
Filipendulosa	*Filipendula ulmaria* (L.), *Urtica dioica* (L.), *Geum rivale* (L.), *Paris quadrifolia* (L.), *Caltha palustris* (L.), *Solanum dulcamara* (L.)
Oxalidosa turf. mel.	*Cirsium oleraceum* (L.) Scopoli, *Hylocomium splendens* (H.) Schimper, *Rhytidiadelphus*, *Brachythecium*, *Vaccinium myrtillus* (L.), *Dryopteris filix-mas* (L.) Schott

During the collection of the empirical data (from October 2019 to June 2021), the air temperature in the study sites ranged from 8.0 ± 0.7 °C to 31.4 ± 0.1 °C (mean 9.2 ± 0.8 °C) and the annual precipitation ranged from 472 mm to 860 mm (average 668 ± 136 mm) according to the data provided by the meteorological stations of the Latvian Environment, Geology, and Meteorology Centre (distance from study site less than 30 km).

The scope of the study included 31 forest stands in total, with the dominant tree species being Norway spruce (*Picea abies* (L.) Karsten), silver birch (*Betula pendula* Roth), and black alder (*Alnus glutinosai* (L.) Gärtner) at different stages of stand development (hereafter spruce, birch, and alder, respectively), from clear-cuts to mature stands (Table 2). The results of the individual peat-layer thickness measurement replicates varied from 23 cm to ≥100 cm (mean 75 ± 7 cm) in the undrained sites and from 25 cm to ≥100 cm (mean 54 ± 12 cm) in the drained sites.

Table 2. Characteristics of forests sites.

Parameter	Undrained Forest Sites				Drained Forest Sites			
	Spruce	Birch	Alder	Clearcut	Spruce	Birch	Alder	Clearcut
Number of study sites	1	3	5	1	12	3	2	4
Age, years	67	21–77	10–80		14–86	18–60	26–53	
Diameter, cm	31	12–29	4–23		2–27	9–27	17–24	
Height, m	28	12–28	4–29		2–24	9–22	17–26	
Basal area, m^2 ha^{-1}	61	17–71	8–57		8–72	19–60	32–56	
Growing stock, m^3 ha^{-1}	335	78–365	35–325		7–521	38–210	123–254	
Thickness of peat layer, cm	68	31–52	30–99	47	37–99	25–75	60–70	63–99

The table shows the range of characteristics of forest sites.

2.2. Sampling and Laboratory Analysis of Soil and Soil Flux

Soil CO_2 flux monitoring was conducted using the manual closed static nontransparent chamber method [19] for 12 consecutive months. Chamber collars were installed at a depth of 5 cm in 5 replicates in each study site. During collar installation, root damage and disturbance of the litter layer were avoided as much as feasible, and GV was left intact throughout the whole monitoring period. Therefore, the flux monitoring represents the CO_2 exchange between the soil surface (including vegetation enclosed in the chamber) and the atmosphere, the sum of soil heterotrophic respiration, and the autotrophic respiration of roots and aboveground ground vegetation (Rfloor), respectively. Soil flux was sampled with an interval of 4 weeks from the chambers in each of the collar positions immediately and at 10, 20, and 30 min after positioning the chambers on the collars. The samples were collected using underpressurized (0.3 mbar) glass vials and tested using the gas chromatography method [20]. The atmosphere and soil temperature at a 5 cm depth (Ts), as well as the groundwater level (using a PVC pipe installed up to a depth of 140 cm), were recorded during the soil flux sampling.

The soil samples were collected with 100 cm^3 cores from fixed soil depths of 0–10 and 10–20 cm in 2 replicates [21]. The soil samples were prepared according to LVS ISO 11464:2005, and the bulk density was determined according to LVS ISO 11272:2017. The soil chemical parameters were determined using standard methods (Table 3). The content of organic C was calculated by subtracting the value of carbonate C from the total C value. In addition, the soil organic C/total N ratio (C/N ratio) was calculated as a proxy to characterize the decomposition of soil organic matter.

2.3. Estimation of Soil Respiration

The acquired analysis results of the CO_2 concentration in the chambers during soil flux sampling were used to calculate the slope values of the linear regression equations characterizing the gas concentration changes over time. The instantaneous Rfloor was calculated using the following equation:

$$Rfloor = \frac{M\ P\ V\ slope}{R\ T\ A} \quad (1)$$

where Rfloor is the instantaneous Rfloor, µg CO_2 m^2 h^{-1}; *M* is the molar mass of CO_2, 44.01 g mol^{-1}; *R* is the universal gas constant, 8.314 m^3 Pa K^{-1}·mol^{-1}; *P* is the assumption of air pressure inside the chamber, 101,300, Pa; *T* is the air temperature, K; *V* is the chamber volume, 0.063 m^3; slope is the CO_2 concentration changes over time, ppm h^{-1}; and *A* is the collar area, 0.1995 m^2.

Table 3. Standard methods used in analyzing soil samples.

Parameter	Unit	Method Principle	Standard Method
Bulk density	kg m^{-3}	Gravimetry	LVS ISO 11272:2017
Total C	g kg^{-1}	Elementary analysis (dry combustion)	LVS ISO 10694:2006
Total N	g kg^{-1}	Elementary analysis (dry combustion)	LVS ISO 13878:1998
Carbonate ($CaCO_3$)	g kg^{-1}	Volumetry	LVS EN ISO 10693:2014
pH	unit	Potentiometry	LVS ISO 10390:2021
HNO_3 extractable potassium (K), calcium (Ca), magnesium (Mg) and phosphorus (P)	g kg^{-1}	ICP-OES	LVS EN ISO 11885:2009

The annual Rfloor was estimated by summing the calculated hourly Rfloor. We calculated the hourly Rfloor by interpolating the measured instantaneous Rfloor using the R_{10} and Q_{10} parameters [22–24], the relationship between the atmospheric temperature and Ts evaluated within the study and the hourly average air temperature data from the nearest meteorological stations. The hourly Rfloor was calculated using the following equation:

$$Rfloor = R_{10} Q_{10}^{\frac{T_S - 10}{10}} \tag{2}$$

where Rfloor is the hourly Rfloor, kg CO_2 ha^{-1} h^{-1}; R_{10} is the Rfloor at a soil temperature of 10 °C, kg CO ha^{-1} h^{-1}; Q_{10} is the temperature sensitivity; and T_S is the soil temperature, °C.

The following equation (R^2 = 0.81, $p < 0.001$), elaborated by the results from previous studies [25], was used to recalculate the annual Rfloor to soil heterotrophic respiration (Rhet):

$$\ln(Rhet) = 1.22 + 0.73 \ln(\mathrm{Rs}) \tag{3}$$

where Rhet is soil heterotrophic respiration, t CO_2 ha^{-1} $year^{-1}$, and Rs is soil respiration, t CO_2 ha^{-1} $year^{-1}$.

The annual Rfloor and Rhet were estimated by stratifying the empirical data acquired in the study according to soil moisture regime (undrained and drained), forest land status (forest stand or clear-cut), and forest type (deciduous or coniferous) to allow the application of study the results for the improvement of the national GHG inventory.

2.4. Estimation of Soil C Input by Litter

The LF samples for the estimation of the annual LF biomass were collected using five conically shaped litter traps (surface area 0.5 m^2) installed in each study site according to the manual methods and criteria for harmonized sampling, assessment, monitoring, and analysis of the effects of air pollution on forests, prepared on behalf of the Programme Co-ordinating Centre and Task Force of ICP Forests [26]. The samples were collected for 12 consecutive months with an interval of 4 weeks.

Separate above- (aGV) and below-ground ground vegetation (bGV) samples were collected in 4 replicates from 20 cm × 20 cm square fields in each study site at the end of the vegetation season when the vegetation biomass had peaked [23]. The FR production samples were collected using the modified ingrowth core method based on a flexible polyester cylindrical bag (diameter 35 mm) with a mesh size of 2 cm × 2 cm installed 60 cm deep in the soil in three replicates in each study site [27,28]. Mesh bags were installed in autumn and removed from the soil after a year by cutting the roots around the bag. The roots of trees were removed from the collected bGV samples, while the roots of GV were removed from the collected FR samples. The soil particles from both types of below-ground biomass samples were removed by wet sieving.

The litter sample dry matter was determined by oven drying (70 °C) the samples; the C content was analyzed by dry combustion using an element analyzer according to LVS ISO 10694:2006. It was assumed that the biomass of the collected foliar litter, GV, and FR production was equal to the annual mortality and respective soil C input:

$$\mathrm{Cinput} = \frac{\mathrm{m} \times 10000 \times \mathrm{C}}{\mathrm{S} \times 100} \quad (4)$$

where Cinput is the annual soil C input by litter, t ha^{-1} year^{-1}; m is the dry matter of litter, t; C is the C content of litter, %; and S is the area of the litter sampler (cross-sectional area of root ingrowth bag, area of LF trap, and area of GV collection field), m^2.

2.5. Estimation of Forest Soil Annual CSC

The soil CSC was calculated as the sum of the soil C input by annual biomass mortality (LF, GV, FR, mosses, and shrubs) and soil C loss by Rhet. The estimated soil CSC was expressed as the mean annual CSC within 240 years of forest management in a business-as-usual scenario. Assumptions of yearly stand age and basal area development within a period of 240 years of forest management (including the impact of harvesting), which were used as variables for the annual soil CSC calculations (Figure 2), are based on the National Forest Inventory data and national stand growth models [29–32].

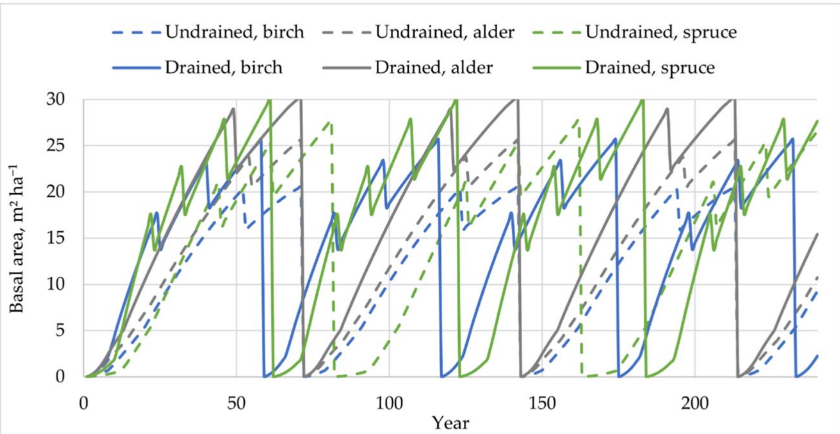

Figure 2. Assumptions of stand basal area dynamics within a 240-year forest management cycle.

The assumptions of the yearly dynamics of the basal area were used as variables for the calculation of the annual soil C input according to the study results. The annual LF C input was calculated using equations for the relationship between the basal area and C content in the annual LF biomass, while the annual soil C input by FR and GV was estimated according to forest land status (forest stand or clear-cut). It was assumed that forest stand or clear-cut status could be determined by the national stand basal area thresholds identifying unproductive stands: 6, 4, and 5 m^2 ha^{-1} for spruce, birch, and alder stands, respectively.

The study results for soil C input and Rhet are supplemented by data on the annual litter biomass of dwarf shrubs and mosses. The biomass of shrubs and mosses was calculated using the equations elaborated within a study conducted in boreal forests [33]:

$$B(spruce)_{shrubs} = \left(10.375 - 0.033A + 0.001A^2 - 0.000004A^3)^2 - 0.5\right) \quad (5)$$

$$B(broadleves)_{shrubs} = \left(7.102 - 0.0004A^2)^2 - 0.5\right) \quad (6)$$

$$B(spruce)_{mos} = \left(19,282 + 0.164A - 0.000001A^3)^2 - 0.5\right) \quad (7)$$

$$B(broadeleves)_{mos} = \left(13.555 + 0.056A)^2 - 0.5\right) \quad (8)$$

where B(spruce)$_{shrubs}$ and B(broadleaves)$_{shrubs}$ are the aboveground biomass of shrubs in coniferous stands and broadleaves forests (kg ha^{-1}), respectively. B(spruce)$_{moss}$ is the aboveground biomass of moss in coniferous forests (kg ha^{-1}), and B(broadleaves)$_{moss}$-is the aboveground moss biomass in deciduous forests (kg ha^{-1}). A is stand age (years).

The annual soil C input by shrubs and mosses was calculated with the assumption that the share of C in the biomass was 47.5% [34] and by multiplying the biomass values with a turnover rate of 0.25 and 0.33 for shrubs and mosses, respectively [33]. It was assumed that 70% of the total C input by dwarf shrubs and mosses contributed to belowground biomass mortality [35–37].

2.6. Statistical Analysis

Statistical analyses were carried out using R (R version 4.0.3; RStudio version 2022.07.1 + 554). A Mann–Whitney U test was used to compare the differences between the two data groups. The correlations were tested with Spearman's ρ. A significance level of $\alpha = 0.05$ was applied in all the tests. The uncertainty of the study results was expressed with confidence intervals ($\alpha = 0.05$).

3. Results

3.1. Soil Characteristics of the Study Sites

The mean organic C content in the top 20 cm of the soil in the studied stands with drained soil was 48.7 ± 4.0% and 45.5 ± 4.3% in stands with undrained soil. Thus, the soil in the studied stands complies with the definition of organic soil [38]. The mean soil C/N ratios in the drained and undrained soil were 19.4 ± 2.8 and 19.2 ± 2.9, and the soil bulk densities were 420 ± 40 kg m^{-3} and 435 ± 43 kg m^{-3}, respectively.

3.2. Soil Respiration

During the soil CO_2 flux monitoring period, the measured Ts ranged from −1.3 to 22.3 °C, while the instantaneous Rfloor ranged from 0.6 to 97.8 µg C m^{-2} s^{-1} (Figure 3). The highest mean instantaneous emissions were found in clear-cuts. The difference between the measured mean Rfloor in the clear-cuts with drained (31.5 ± 7.0 µg C m^{-2} s^{-1}) and undrained (33.4 ± 14.4 µg C m^{-2} s^{-1}) soil is not significantly different. The measured mean Rfloor in the forest stands with different dominant tree species, and the soil moisture regimes were also not significantly different from each other and ranged from 18.5 ± 7.1 µg C m^{-2} s^{-1} in spruce stands with undrained soil to 25.9 ± 7.2 µg C m^{-2} s^{-1} in birch stands with drained soil. However, the difference between the measured mean Rfloor in the clear-cuts (31.9 ± 62 µg C m^{-2} s^{-1}) and forest stands (21.7 ± 1.8 µ C m^{-2} s^{-1}) is significantly different.

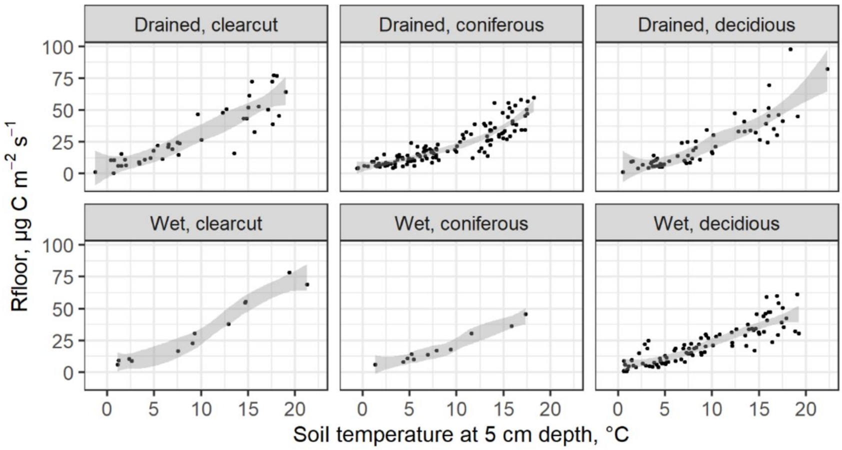

Figure 3. Relationship between soil temperature and Rfloor. Confidence intervals are shown around the smooth lines.

The relationship between the Ts and Rfloor can be expressed by exponential regression. The equation (Rfloor = a × exp (b × Ts) coefficients a and b are summarized in Table 4. According to the study results, the Rfloor tended to be more sensitive to Ts changes in deciduous forests and drained clear-cuts (Q_{10} from 3.9 to 4.18) as compared to the other study sites, namely, clear-cuts with undrained soils and coniferous forests (Q_{10} from 3.25 to 3.46).

Table 4. Summary of the models for the prediction of soil Rfloor by soil temperature at a depth of 5 cm.

Moisture Regime	Forest Site Characteristics	Characteristics of Measured Data (Range)		Model Coefficients		Model Characteristics	
		Ts, °C	Rfloor, µg C m^{-2} s^{-1}	a	b	RMSE	Q_{10}
Drained	Clear-cut	−1.3 … 9.0	0.5 … 77.2	5.784	0.141	11.9	4.10
	Deciduous	0.5 … 22.3	1.2 … 97.8	4.476	0.143	10.3	4.18
	Coniferous	−0.6 … 18.2	4.2 … 59.7	6.235	0.118	6.2	3.25
Undrained	Clear-cut	1.1 … 21.3	6.3 … 78.5	7.298	0.124	10.6	3.46
	Deciduous	0.7 … 19.3	0.6 … 61.3	4.700	0.136	8.5	3.90
	Coniferous	1.4 … 17.4	6.2 … 45.9	5.798	0.124	3.0	3.46

The annual Rfloor was estimated by applying the prediction models developed by the study and hourly air temperature data within the study period in combination with the observed relationship between the temperature of the atmosphere and Ts (R^2 = 0.87, $p < 0.05$):

$$\mathrm{Ts} = 0.715 t_{air} + 1.719 \quad (9)$$

where *Ts* is the soil temperature at a 5 cm depth, °C, and t_{air} is the air temperature, °C. The estimated annual Rfloor ranged from an average of 5.1 ± 0.2 t C ha^{-1} $year^{-1}$ and 5.1 ± 2.6 t C ha^{-1} $year^{-1}$ in alder stands with drained soil and birch stands with undrained soils, respectively, to 7.9 ± 3.3 t C ha^{-1} $year^{-1}$ in clear-cut stands with undrained soil. The estimated annual mean Rfloor in the forest sites and clear-cuts was 6.2 ± 0.4 t C ha^{-1} $year^{-1}$

and 7.7 ± 1.7 t C ha^{-1} $year^{-1}$, respectively. The empirical data acquired show a correlation between the Rfloor and GV biomass (r was 0.4 to 0.55 for bGV and GV, respectively). It was also observed that the soil C content (r = 0.51) and LF biomass (r = −0.59) had a moderate correlation with Rfloor, while stand age had a weak (r = −0.36) but significant ($p < 0.05$) impact on Rfloor (Figure 4).

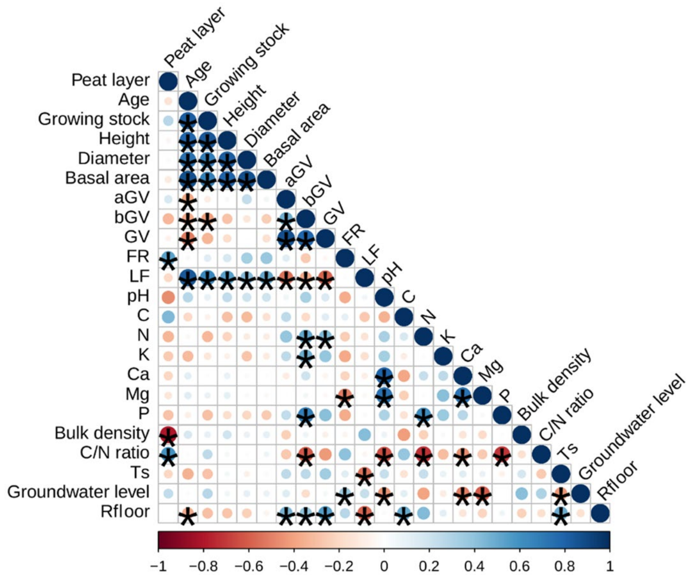

Figure 4. Correlation analysis of the soil C stock balance components and affecting factors. Size and color of the bubbles indicate correlation strength; starred bubbles show significant ($p \leq 0.05$) correlations.

3.3. Soil C Input by Litter

In the study sites, the aGV (mean C content 47.1 ± 0.7%) and bGV (mean C content 49.7 ± 0.8%) at the end of vegetation season ranged from 6.1 to 8.2 t ha^{-1} (average 6.9 ± 1.0 t ha^{-1}) in clear-cuts to 1.3 to 6.5 t ha^{-1} (average 3.5 ± 0.7 t ha^{-1}) in forest stands. While there was no statistically significant relationship identified between the aGV and soil chemical parameter data, the bGV data had a relationship with the parameters indicating soil fertility. The bGV data had a moderate correlation with soil N (r = 0.51), K (r = 0.49), P (r = 0.69) content, and C/N ratio (r = −0.62). Although GV had a moderate correlation (r = 0.51) with stand age, due to the lack of study data available to elaborate models based on stand variables, soil CSC modeling was chosen to be performed by fixed ground vegetation biomass values stratified according to forest land status (forests stand or clear-cut), moisture regime (drained or undrained soil), and dominant tree species (coniferous or deciduous).

The same approach was applied regarding the FR litter data. The estimated FR production in the forest stands ranged from 0.1 to 1.8 t ha^{-1} (average 0.8 ± 0.2 t ha^{-1}). Although moderate correlations between the estimated FR production and soil fertility characteristics data exist, these relationships were not found to be significant, except in the case of soil Mg content. The study data show that a lower annual average groundwater level tended to increase FR production (r = 0.38).

The estimated annual LF biomass ranged from 0.5 to 5.7 t ha^{-1} (average 3.3 ± 0.5 t ha^{-1}) with a mean C content of 52.1 ± 0.2%. The annual foliar litter biomass data had a moderate (r from 0.44 to 0.65) correlation with average tree diameter, basal area, height, and growing stock (in order of increasing correlation) to a high correlation with stand age (r = 0.84). The basal area was chosen as a predictor for the explanation of the annual foliar litter biomass due to its better representation of the impact of deciduous or coniferous tree species. The annual soil C input by litter in the spruce stands had a linear relationship with the stand basal area; in the study sites, the estimated C input increased from 0.26 to 2.34 t C ha^{-1} year^{-1} in stands with a basal area of 8 m^2 ha^{-1} and 45 m^2 ha^{-1}. The acquired data suggest that, in the case of deciduous forests, the LF stands had a steeper biomass increase until the basal area reached around 20 m^2 ha^{-1}. When the annual C input by litter reaches around 1.5 t C ha^{-1} year^{-1}, further increases in the basal area have a more gradual impact on litter biomass increases (Figure 5).

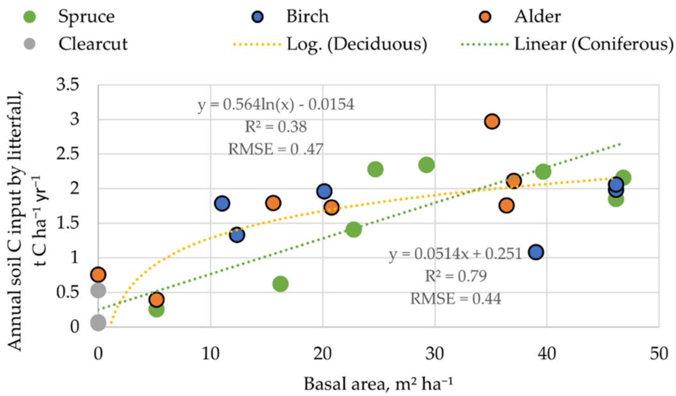

Figure 5. Relationship between C stock in annual foliar litter and stand basal area.

3.4. Summary of Estimated Annual Soil CSC

According to the empirical data of the annual average Rfloor recalculated to Rhet and the soil C input estimated in the study, summarized in Table 5, the soil C loss by Rhet is compensated by the annual soil C input in forest stands with drained and undrained soils, while drained and undrained soil in clear-cuts is a net CO_2 source.

Table 5. Summary of estimated soil respiration and soil C input (t C ha^{-1} year^{-1}) in the study sites.

Moisture Regime	Forest Site Characteristics	Rfloor	Rhet	aGV	bGV	FR	LF	Net Balance
Drained	Clear-cut	−7.6 ± 2.2	−4.3 ± 0.90	2.2 ± 0.3	1 ± 0.2		0.2 ± 0.2	−0.9 ± 0.7
	Deciduous	−6.2 ± 1.2	−3.7 ± 0.53	0.7 ± 0.3	1.3 ± 0.3	0.3 ± 0.3	1.6 ± 0.3	0.2 ± 0.4
	Coniferous	−6.3 ± 0.5	−3.7 ± 0.23	0.8 ± 0.2	0.8 ± 0.2	0.4 ± 0.1	1.7 ± 0.5	0.0 ± 0.3
Undrained	Clear-cut	−7.9 ± 1.2	−4.4 ± 0.50	2.4 ± 0.3	1.4 ± 0.3		0.1 ± 0.4	−0.4 ± 0.4
	Deciduous	−6.1 ± 0.7	−3.7 ± 0.32	0.7 ± 0.3	1.0 ± 0.4	0.3 ± 0.1	1.7 ± 0.6	0.0 ± 0.4
	Coniferous	−5.1 ± 1.2	−3.3 ± 0.50	0.5 ± 0.3	0.8 ± 0.3	0.5 ± 0.1	2.2 ± 0.4	0.8 ± 0.4

According to the modeling exercise explained in Section 2.4., within a 240-year forest management cycle, the annual soil CSC ranged from −1.0 to 2.6 t C ha^{-1} year^{-1} (mean 0.4 t C ha^{-1} year^{-1}) in deciduous forests and from −0.6 to 2.9 t C ha^{-1} year^{-1} (mean 0.7 t C ha^{-1} year^{-1}) in coniferous forests with undrained soil, whereas in forests with drained soil, the annual net C balance ranged from −1.3 to 1.0 t C ha^{-1} $year^{-1}$ (mean 0.1 t C ha^{-1} $year^{-1}$) in stands with deciduous-dominant species and from −1.3 to 0.1 t C ha^{-1} $year^{-1}$ (mean −0.6 t C ha^{-1} $year^{-1}$) in stands with coniferous-dominant species (Figure 6). These results indicate that long-term drainage reduces C uptake by nutrient-rich organic soil in managed forests. After drainage, the soil in deciduous forests may remain C-neutral, but in coniferous forests, the soil may become a CO_2 source.

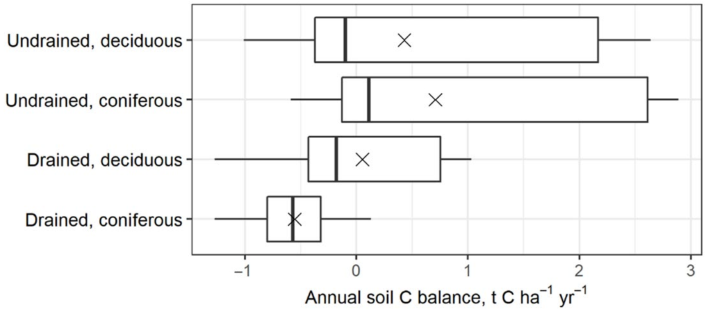

Figure 6. Inter-annual soil CSC variation within a 240-year forest management cycle. In the boxplots, the median is shown by the bold line; the mean is shown by the cross; the box corresponds to the lower and upper quartiles; whiskers show the minimal and maximal values (within 150% of the interquartile range from the median).

The soil CSC modeling results showed that aboveground litter, on average, contributed to the annual soil C input by 60 ± 4%, of which 40 ± 10% was from LF and 57 ± 9% from aboveground ground vegetation. The main component of the soil C input was GV, contributing 60 ± 7% of the total annual soil C input (Table 6).

Table 6. Summary of estimated forest management cycle's annual average soil CSC (t C ha^{-1} $year^{-1}$) for the study site measurements.

Soil CSC Component	Drained		Undrained	
	Deciduous	Coniferous	Deciduous	Coniferous
Rhet	−3.82 ± 0.45	−3.85 ± 0.36	−3.88 ± 0.7	−3.54 ± 1.38
LF	1.14 ± 0.21	0.81 ± 0.24	1.12 ± 0.46	0.73 ± 0.22
FR	0.31 ± 0.25	0.44 ± 0.13	0.32 ± 0.29	0.53 ± 0.16
aGV	0.93 ± 0.61	1.1 ± 0.31	1.79 ± 0.61	1.59 ± 0.45
Aboveground shrubs	0.01 ± 0.01	0.01 ± 0.01	0.01 ± 0.01	0.01 ± 0.01
Aboveground mosses	0.02 ± 0.02	0.09 ± 0.04	0.02 ± 0.02	0.11 ± 0.04
bGV	1.27 ± 0.42	0.59 ± 0.17	1.13 ± 0.52	1.00 ± 0.28
Belowground shrubs, mosses	0.07 ± 0.07	0.25 ± 0.1	0.07 ± 0.07	0.28 ± 0.12
Net balance	−0.08 ± 0.03	−0.55 ± −0.12	0.43 ± 0.17	0.71 ± 0.25

4. Discussion

4.1. Method of Rhet Calculation

Previous studies in boreal forests mainly focused on the direct evaluation of Rhet [8]. In our study, the decision to estimate soil respiration by Rfloor measurements is a result of a methodological compromise allowing the acquisition of collected gas sample analysis results for soil CH_4 and N_2O flux estimates as well [39]. For this reason, in our study, the

Rfloor measurements were recalculated to Rhet using an equation elaborated from the data of previous studies. While such an approach may introduce additional uncertainty in the results of Rhet estimates, it allowed us to maintain the simplicity of the soil CSC calculation. Although the use of direct Rhet measurement results would avoid such unknown potential errors, the method (trenching) required to acquire such Rhet data may introduce other errors in the flux estimates due to altered soil conditions [40–42]. The soil CSC calculation method based on the Rhet data was also favored by a study that compared different soil CSC estimation methods based on chamber measurements against reference estimates using eddy covariance measurements. In this study, it was concluded that an approach based on Rhet data provided results that agreed better with the reference results, as compared to methods that use Rfloor measurement data [25]. It was found that, although both soil CSC estimation methods are sensitive to biases introduced by the soil C input and output data used in the calculations, the approach based on Rhet data was more applicable due to the relatively simple calculation approach of subtracting Rhet measurement results from the soil C input data, while complicated modeling of ecosystem photosynthesis and respiration is needed to calculate soil CSC using Rfloor measurement data.

To recalculate the study results of the Rfloor for individual study sites to Rhet, a factor ranging from 0.52 to 0.66 (mean 0.6) was used. Therefore, the calculated Rhet may be overestimated, as the equation applied to determine the Rhet/Rfloor recalculation factor was elaborated by comparing the data of Rhet and Rfloor, excluding aboveground autotrophic respiration (Rs), from studies conducted in both boreal and temperate zones [25]. However, such an assumption may be speculative as, according to the data compiled by more recent metanalytical reviews, the Rhet/Rs determined by the trenching method in boreal coniferous forests ranges from 0.36 to 1.03 (mean: 0.73, with a standard deviation of 0.18). Therefore, the approach used to calculate the Rhet in the study may be considered conservative from the GHG inventory perspective as underestimation of soil C loss is not favorable in the elaboration of soil emission factors.

The results of Rhet calculated for individual study sites with forest cover ranging from 2.9 to 4.4 t C ha^{-1} $year^{-1}$ fall within the range of results of the Rhet estimated by direct measurement in other studies in boreal forests. The Rhet of forestry-drained peatlands reported in the results of a Finnish study ranges from 1.46 to 6.70 t C ha^{-1} $year^{-1}$. [16]. The Rhet estimated in a 30-year-old Scots pine plantation (former cropland) with organic soil situated in the middle of a boreal climatic zone was 4.80 t C ha^{-1} $year^{-1}$ [43], while the quantified Rhet of 12 afforested organic soil cropland sites and six cutaway peatlands in Finland ranged from 2.07 to 5.39 t C ha^{-1} $year^{-1}$ and from 2.76 to 4.79 t C ha^{-1} $year^{-1}$, respectively [44]. The results of another study carried out in Finland showed an average Rhet of 2.38 t C ha^{-1} $year^{-1}$ in a pine-dominated drained mire. It was estimated that the annual Rhet of forestry-drained peatlands in central Estonia and southern and northern Finland ranges from 2.48 to 5.15 t C ha^{-1} $year^{-1}$ [13]. The consistency of the Rhet estimated in our study with previously reported values indicates that the use of the Rfloor recalculation method is applicable for studies conducted in the hemiboreal zone.

4.2. Soil Respiration

In our study, the difference between the mean measured instantaneous Rfloor in drained sites (7.35 ± 0.89 t C ha^{-1} $year^{-1}$) and undrained (7.02 ± 0.96 t C ha^{-1} $year^{-1}$) forest stand sites were found to be insignificant (p = 0.34). However, the differences between the measured mean instantaneous Rfloor in forest stands (6.84 ± 0.56 t C ha^{-1} $year^{-1}$) and clear-cuts (10.08 ± 1.96 t C ha^{-1} $year^{-1}$) were significant (p = 0.002). The tendency of similar soil respiration in drained and undrained sites, as well as increased emissions in areas with no forest cover, was also observed in a previous study. For instance, in afforested lowland raised peat bogs in Scotland, it was found that Rfloor was slightly higher in drained sites (4.53 t C ha^{-1} $year^{-1}$) compared to undrained sites (3.35 t C ha^{-1} $year^{-1}$), while in undrained areas with no forest cover, the estimated Rfloor was 6.95 t C ha^{-1} $year^{-1}$ [45].

In our case, the insignificant difference between average soil respiration in drained and undrained study sites is mainly determined by the absence of correlation between the measurement data of the Rfloor and groundwater level. The average groundwater level in the drained study sites (mean 55 ± 2 cm) was on average 18 ± 2 cm deeper than in the undrained sites (mean 35 ± 3 cm); however, the groundwater level measurement results have weak, insignificant correlation (r = 0.3) with soil respiration data. We were not able to find empirical reasons for having a significantly higher soil respiration rate in clear-cut study sites compared to sites with forest cover. By evaluating the data of Ts and atmospheric temperature measurements, we observed that there was no higher Ts sensitivity to variation in atmospheric temperature. Linear regression models of characterizing the relationship between Ts and soil temperature in forest stands and clear-cuts were not statistically significant. Therefore, we concluded that Ts response to changes in atmospheric temperature was not different in both study site groups and increased warming of soil in clear-cuts was not the reason for elevated soil respiration. Most likely, the increased emissions are induced by soil disturbances of mechanized harvesting [46] and the decomposition of harvesting residues [47].

The annual Rfloor in clear-cuts with peaty gley soil, estimated by the previous study as ranging from 6.5 ± 1.6 to 7.1 ± 1.7 t C ha^{-1} $year^{-1}$ [48], which is similar to our estimation of the annual Rfloor in drained and undrained clear-cuts, i.e., 7.6 ± 2.2 and 7.9 ± 1.2 t C ha^{-1} $year^{-1}$, respectively. Additionally, the estimated Rfloor in the study sites with forest cover, which ranged from 4.4 to 8.0 t C ha^{-1} $year^{-1}$, is similar to the range of the Rfloor estimated in other studies (2.73 ± 0.55 to 5.18 ± 1.09 t C ha^{-1} $year^{-1}$) conducted in boreal forests [49,50]. Furthermore, the mean Rfloor was found to be significantly higher in drained coniferous forests with organic soil (from 2.45 to 5.18 t C ha^{-1} $year^{-1}$) than in undrained mire forest sites (from 2.18 to 3.27 t C ha^{-1} $year^{-1}$), although the drained sites were all moist [51]. This may be in line with the observations made in our study that the groundwater level may have no significant impact on Rfloor. Additionally, in a study aimed at creating soil respiration prediction models, it was concluded that by adding the water table depth into the models as an explanatory variable, the goodness of fit was not improved and the prediction power was not statistically significantly improved [52]. Even though, in some cases, the average water table depth can be significantly correlated with annual respiration values in peatlands [53], soil temperature alone is generally sufficient to explain the variation in soil respiration. The reasons why groundwater level can be used as a Rfloor predictor only in some areas can be further studied.

4.3. Soil C Input by Litter

The study results of the average annual soil C input by LF in drained and undrained forests with nutrient-rich organic soils ranging from 1.6 ± 0.3 to 2.2 ± 0.4 C ha^{-1} $year^{-1}$, respectively, are within the uncertainty range of the average observed values in the coniferous and deciduous forests of Northern Europe, 1.7 ± 1.1 and 1.5 ± 0.7 t C ha^{-1} $year^{-1}$, respectively [54]. While similar relationship tendencies with the basal area have been recognized, higher estimated values of the average soil C input by LF of coniferous (1.82 ± 0.02 t C ha^{-1} $year^{-1}$) and silver birch stands (2.07 ± 0.03 t C ha^{-1} $year^{-1}$) with drained organic soils were found in a recent Latvian study [55]. This points out that the average soil C input values used in the estimates of forest C balance or comparison of litterfall biomass across different studies can lead to considerable inaccuracies. In our study, as well as in a previous local study [55], it is recognized that basal area provides the highest prediction power of litterfall biomass compared to other commonly used forest stand characteristics. Therefore, the variation in the LF data acquired in the study can be explained with the basal area of the forest stands studied. A limited number of study sites restricted the ability to compare the relationship between the basal area and LF in the drained and undrained sites separately.

The mean tree fine-root production, ranging from 0.6 ± 0.6 t ha^{-1} $year^{-1}$ in drained deciduous forests to 1.0 ± 0.2 t C ha^{-1} $year^{-1}$ in undrained coniferous forests with nutrient-rich organic soil, as estimated in this study, is significantly lower than those mostly reported in previous studies. The mean fine-root production reported for Northern Europe was 2.84 ± 1.52 t ha^{-1} year^{-1} in coniferous forests and 1.99 ± 1.01 t ha^{-1} year^{-1} in deciduous forests [56]. Lower estimated fine-root production values may be explained by methodological underestimation or the phenomenon whereby the growth of trees in nutrient-rich soil requires less biomass of the fine roots to ensure a sufficient intake of water and nutrients. Higher fine-root productivity in stands with less fertile soils has been noticed in several studies [22,57–59]; however, the opposite relationship has also been found [60]. Such assumptions may also be contradicted by the annual fine-root production in forests with drained nutrient-rich soil, which ranged from 1.81 to 3.02 t ha^{-1} year^{-1}, as reported in an Estonian study [12]. Most likely, the reason for underestimation arises from methodology, as the average uncertainty of acquired results also ranges from 30 to 161 % (mean 71 %) in study sites with different dominant tree species and soil moisture regimes. The study period of one year was not sufficient for fine root production estimates by the ingrowth method, as one vegetation season was not sufficient to mitigate the effects of disturbance introduced by the installation of ingrowth bags.

The annual soil C input by GV has not been studied extensively, and the available results are often not comparable due to different study methods and the different vegetation components included in the calculations. In Estonia, the estimated primary production of dwarf shrubs and grasses reached 0.4 t C ha^{-1} year^{-1} in spruce stands and ranged from 0.6 to 1.0 t C ha^{-1} year^{-1} in pine stands [22]. By using the biomass of herbs and grasses, the prediction models elaborated by a study conducted in Finland [33], taking into account the age distribution of Latvian forests, the average weighted annual soil C input by aGV and bGV ranged from 0.34 ± 0.01 to 1.29 ± 0.202 t C ha^{-1} year^{-1} in birch and pine stands with drained organic soil, respectively [55]. The higher annual soil C input by herbs and grasses, which ranged from 0.6 to 3.2 t C ha^{-1} year^{-1} in forest stands and from 2.9 to 4.0 t C ha^{-1} year^{-1} in clear-cuts estimated in our study, can be explained by the forest site types characterized by nutrient-rich soils included in this study and the positive correlation found between GV biomass and soil fertility characteristics.

4.4. Annual Net Soil CSC

According to general opinion, the drainage of organic soil increases CO_2 emissions and reduces soil C stock; however, the results of previous studies on the effect of organic soil drainage on GHG emissions are ambiguous. The empirical data collected during this study shows that nutrient-rich organic soil in forest stands is a net CO_2 sink, but the soil in clear-cuts is a net CO_2 source. We estimated that during the study period, the average annual soil CSCs were 0.4 ± 0.4 t C ha^{-1} year^{-1} in undrained and 0.1 ± 0.4 t C ha^{-1} year^{-1} in drained forest sites, while in clear-cut estimated soil net C balance is −0.9 ± 0.7 and −0.4 ± t C ha^{-1} year^{-1} in drained and undrained sites, respectively. The observation of soil in forest stands acting as a C sink is in agreement with the conclusion reached in a previous local study on the CSC of drained moderately nutrient-rich forest soils [14] and can be explained by site productivity induced increased C input by litter that fully compensates soil C loss by respiration. The reason for soil in clear-cuts being a C source was increased C loss by respiration and reduced soil C input by litter. Although mean soil C input by ground biomass in clear-cuts (3.55 ± 0.37 t C ha^{-1} year^{-1}) was considerably higher than in forest stands (1.65 ± 0.40 t C ha^{-1} year^{-1}), that did not compensate for the increase in Rhet by average 0.8 t C ha^{-1} year^{-1} compared to forest sites and the absence of C input by litterfall (average 1.8 ± 0.5 t C ha^{-1} year^{-1}) and the fine roots of trees (0.4 ± 0.2 t C ha^{-1} year^{-1}).

In addition to calculating the annual soil CSCs as a sum of soil C balance components quantified in a monitoring year of the study, we modeled an inter-annual soil CSC within a 240-year forest management period by using variables of stand characteristics as predictors. As a result, by taking into account the impact of forest stand development (age and basal area) and the stages of forest land (forest stand and clear-cut), the average soil

CSC with high variability was estimated. The range of estimated annual soil CSCs is mainly determined by two soil C balance components with high inter-annual variability determined by the stage of forest stand development—LF and ground GV. The most significant impact on variability is introduced by GV, which determines on average 63% of the total soil C input by litter, while LF impacts on average 24% of annual soil C input. Empirical data with the highest uncertainty is soil C input by FR, aGV, and bGV with confidence intervals of 71%, 41%, and 37%. The annual soil C input by GV ranges from an average of 1.68 ± 0.36 to 4.88 ± 1.40 t C ha^{-1} year in clear-cuts and forest stands with various dominant tree species signifying the importance of reducing the uncertainty of annual soil CSC estimations introduced by data on GV.

The results of the study are in line with the previous studies, which showed that the soil C stock does not change and can even increase after the drainage of organic soil in boreal forests [11,24,33,34]. It can be expected that soil respiration may be considerably higher in nutrient-rich site types compared to site types with less fertile soils [13], which may lead forests with fertile drained organic soils to be a source of CO_2 emissions for the following reason. The Rfloor of forestry-drained peatlands estimated in Finland showed a clear diminishing trend in annual soil respiration from the most to the least fertile site types, and ranged from 3.8 to 12.10 t C ha^{-1} year^{-1} [16]. However, the results of this study indicate that both undrained and drained nutrient-rich organic soil in forest stands can still be a net C sink. The differences in the calculated annual soil CSC across various studies may be due to the different methods applied and the inclusion of different soil C input components in the calculations [52], as well as the uncertainty of these values, since the data available, especially for belowground litter, are highly uncertain, most often due to difficulties in acquiring such data [56]. For example, drained forest peatlands were identified as a CO_2 source also in Sweden; however, the estimated annual soil C loss of -2.29 t C ha^{-1} year^{-1}, calculated by subtracting Rhet from the soil C input [61], is considerably higher than that in our study, whereas in Finland, peatlands drained for forestry were found to be a net CO_2 sink (removals from 0.2 to 0.252 t C ha^{-1} year^{-1}) estimated by the soil C inventory method [62]. In our study, the inclusion of forest land status as a clear-cut estimation of annual soil CSC determined if nutrient-rich drained organic forest soil acts as a CO_2 sink or source. The differences highlight the importance of harmonizing soil CSC estimation methods to improve the comparability of country-level GHG inventory results.

The results of soil CSC acquired in this study can be further improved by both more extensive studies and by conducting direct Rhet measurements or evaluating the proportion of Rhet/Rfloor under national conditions. Instead of using static annual soil C input value, approaches to model inter-annual litter biomass variations based on forest stand variables and climatic conditions should be elaborated for the estimation of soil C balance by offsetting the annual Rhet modeled using the annual data of air or soil temperature. The inter-annual variation in hourly or diurnal temperature data may have a significant impact on modeling soil respiration using previously elaborated equations characterizing the relationship between soil respiration and soil or atmosphere temperature. The choice of using temperature data of one year or time period characterizing climate, as well as the use of daily mean or hourly mean temperature data, may have a considerable impact on soil respiration modeling results, which should be considered in future studies.

5. Conclusions

The drained and undrained nutrient-rich organic soils in the forest stands monitored for one year in this study were a CO_2 sink, while the soil in clear-cuts acted as a CO_2 source. The soil in clear-cuts acting as a CO_2 source was determined by increased soil respiration rates and the absence of soil C input by litterfall and the fine roots of trees. The significantly increased soil C input by ground vegetation in clear-cuts mitigated this effect; however, the significantly increased soil respiration rate and reduced soil C input by other sources were not fully compensated. If forest management cycles are considered, including forest land state as a clear-cut, drained nutrient-rich organic soil in managed forests is a CO_2 source, while the soil C stock increases in undrained soil, according to the methodology applied in the CSC calculations.

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Article

Variation in the Mercury Concentrations and Greenhouse Gas Emissions of Pristine and Managed Hemiboreal Peatlands

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Abstract: We assessed total mercury (THg) concentrations and greenhouse gas (GHG) emissions in pristine and managed hemiboreal peatlands in Latvia, aiming to identify environmental factors that potentially affect their variation. The THg concentrations in soil ranged from <1 μg kg^{-1} to 194.4 μg kg^{-1}. No significant differences between THg concentrations in disturbed and undisturbed peatlands were found, however, the upper soil layer in the disturbed sites had significantly higher THg concentration. During May–August, the mean CO_2 emissions (autotrophic and heterotrophic respiration) from the soil ranged from 20.1 ± 5.0 to 104.6 ± 22.7 mg CO_2-C m^{-2} h^{-1}, N_2O emissions ranged from −0.97 to 13.4 ± 11.6 μg N_2O-N m^{-2} h^{-1}, but the highest spatial variation was found for mean CH_4 emissions—ranging from 30.8 ± 0.7 to 3448.9 ± 1087.8 μg CH_4-C m^{-2} h^{-1}. No significant differences in CO_2 and N_2O emissions between disturbed and undisturbed peatlands were observed, but CH_4 emissions from undisturbed peatlands were significantly higher. Complex impacts of environmental factors on the variation of THg concentrations and GHG emissions were identified, important for peatland management to minimize the adverse effects of changes in the biogeochemical cycle of the biophilic elements of soil organic matter and contaminants, such as Hg.

Keywords: mercury; GHG emissions; peatland; peatland management; hemiboreal zone

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1. Introduction

Organic soils, formed where the intensity of decomposition is lower than the production of organic matter, comprise approximately 2% of the ice-free land surface globally, and their majority is wetlands. Relatively pristine organic soils still occur in peatlands in northern European countries, mostly in Norway, Sweden and Finland [1]. Peatlands, which cover 4.23 million km^2 worldwide, are terrestrial ecosystems (a type of wetlands) with or without vegetation that have naturally accumulated at least a 30-cm-thick layer of peat, formed from carbon-rich dead and decaying plant material under permanent water saturation and low oxygen (O_2) conditions. In Europe, more than half of the soil organic carbon (C) stocks are present in peatlands [2]. Historically, a substantial area of peatlands has been drained for production purposes such as agriculture, forestry, grazing and peat extraction [3–5]. In Latvia, drained organic soils comprise 628.6 kha in total, including 425.1 kha of forest land, 39.7 kha of wetlands (peat extraction fields), 76.0 kha of grassland, 78.6 kha of cropland and 9.3 kha of settlements [6].

Peatlands provide many important ecosystem services, including water supply regulation and flood risk mitigation, global biodiversity preservation, climate change mitigation and material for energy production and recreation [4]. Peatlands play an important role in the control of atmospheric greenhouse gases (GHGs) such as carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) and thus affect global climate change [7–9]. The dynamics of C balance and GHG flux in peatlands depend greatly on peatland hydrology [3]. Peatlands usually act as long-term C and GHG sinks [3,9–11]. In the anaerobic zones of submerged soils, CH_4 is produced by methanogens and substantial quantities of C are emitted as CH_4 in the terminal step of anaerobic organic matter mineralization [3,5,12]. Drainage immediately decreases the ground water (GW) level [13], which increases the availability of O_2 and stops anaerobic decomposition and the associated emission of CH_4 by decreasing CH_4 production and increasing the oxidation of CH_4 into CO_2 by methanotrophs [3,12,14–16]. At the same time, aeration results in the aerobic decomposition of peat, releasing CO_2 and N_2O into the atmosphere [17]. Unmanaged peatlands generally emit negligible N_2O [3], but after drainage, N_2O emissions increase, especially in fertile sites such as nitrogen-rich minerotrophic peatlands [3,18,19] due to nitrification, a process that produces nitrate and N_2O as by-products in oxic conditions [16].

The soil becomes a source of net GHG emissions when C and nitrogen (N) loss through organic matter decomposition is not balanced by input via biomass mortality. Soil respiration, especially heterotrophic respiration, is a major cause of soil C loss [20–22], while the main soil C input source is aboveground litter and fine root turnover [23]. Soil respiration and C input are mainly affected by the soil temperature, moisture regime and fertility [14,18,24–26], indicated by the share of organic matter [27] and the C/N ratio [28]. Heterotrophic CO_2 emissions correlate with soil bulk density [18] and chemical composition, which determine the rate of organic matter decomposition [29]. Furthermore, a low C/N ratio increases soil N_2O emissions [30,31], while the soil moisture regime, which is affected by GW level, influences the balance between CH_4 emissions and removal via methanogenesis and methanotrophy [32].

In Latvia, most of the knowledge about the effect of peatland management on GHG emissions and C sequestration is recent and incomplete, especially concerning the most appropriate peatland management measures to mitigate climate change. There is evidence that draining peatlands does not necessarily reduce the soil C sink. Establishing a forest site of *Myrtillosa turf. mel.* with moderate nutrient-rich drained organic soil did not reduce the soil C stock of the former transitional mire, indicating that C stock can increase after drainage due to an increased above- and below-ground litter production [33]. In addition, direct GHG emission measurements with the chamber method revealed a neutral impact of draining nutrient-poor forest organic soils on GHG emissions [34]. In forest sites with nutrient-rich organic soil in Latvia, drained soil is not necessarily a source of CH_4 emissions, while the estimated CH_4 emissions from naturally wet soils are highly variable. Soil becomes a source of CH_4 emissions when the GW level decreases below 20 cm, while the difference in N_2O emissions from drained and naturally wet sites is not significant [35].

Another important environmental issue in the research focus on peatlands is the large store of mercury (Hg) in them that could function as a Hg source for over a century [36] even if deposition of Hg is significantly reduced [37]. Peatlands are often considered biogeochemical hot spots [38,39] of Hg transformation through biotic methylation processes, and they are significant sources of methylmercury (MeHg) to hydrologically connected aquatic ecosystems such as streams and lakes (e.g., [36,40–45]). However, a precondition of higher MeHg concentrations is elevated total Hg (THg) pools in peatlands, mostly resulting from increased atmospheric Hg deposition over the decades [46,47] and the high affinity of soil organic matter (SOM) to Hg, as well as abundant reduced sulphur (S) sites on organic matter molecules that provide strong binding sites for Hg [48]. One of the main pathways of Hg deposition is the wet and dry deposition of oxidized atmospheric Hg (Hg^{2+}) by precipitation directly onto soils or indirectly from plant surfaces via throughfall [49,50]. In terrestrial ecosystems, litterfall has been revealed as the main pathway for the atmosphere–surface transfer of Hg [51,52]. After its deposition through litterfall, biogeochemical reactions limited by different environmental factors determine the further transformation and flow of Hg in ecosystems [52]. Concerns in Latvia have been raised over Hg concentrations in freshwater biota exceeding the threshold of 0.02 mg kg^{-1} (wet weight) set by the national environmental quality standard (Regulations Regarding the Quality of Surface Waters and Groundwaters) [53].

This study sought to compare GHG emissions and the THg concentration in the soil of undisturbed (pristine) and disturbed (managed) peatlands to examine the effect of management and identify the environmental parameters including soil general chemistry and vegetation composition affecting these aspects. In the context of this research, disturbed peatlands are peatlands where anthropogenic influences, such as drainage for agriculture, forestry or peat extraction, have lowered the originally high GW level and changed the vegetation composition. We hypothesize that peatland management is one of the major factors influencing both studied environmental threats—GHG emissions and THg concentration in soil.

2. Materials and Methods

2.1. Research Sites

This study was conducted in 2019 in Latvia (in a hemiboreal zone). In Latvia, the mean annual precipitation in 2019 was 629.2 mm, which is 9% below the annual norm (692.3 mm). The mean annual air temperature in 2019 was +8.2 °C, the minimum mean monthly temperature was −4.0 °C (January 2019) and the maximum mean monthly temperature was 18.6 °C (June 2019) [54].

In total, 22 research sites were selected in peatlands located mostly in central and northern Latvia (Figures 1 and S1–S11). At the research sites, the peat layer thickness was >50 cm.

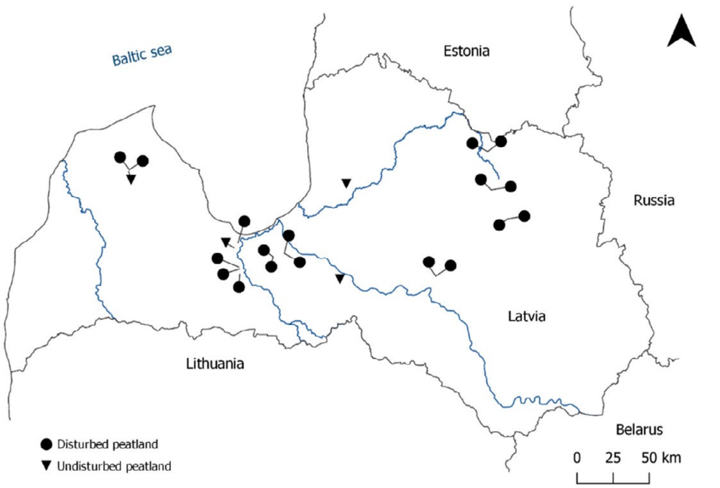

Figure 1. Location of the research sites in Latvia.

In the context of this study, anthropogenic interventions that altered the natural peatland ecosystem, e.g., establishing drainage systems and lowering the GW level, extracting peat, using land for forestry and agricultural purposes and other management practices were considered disturbances. Undisturbed research sites were located in pristine (natural) peatland with no documented management history. In disturbed research sites, drainage systems (ditches) were established, and at most sites, peat had been extracted (Table 1).

Table 1. Characterization of the research sites in Latvia.

Management-Induced Disturbance	Short Description of Management-Induced Disturbance	Current Land Use/ Type of Vegetation	Research Site	Latitude, Longitude	Short Description of Research Site
Undisturbed peatland	Natural (pristine) peatland—undrained, peat have not been extracted previously	Transitional mire	Kalnezers Mire	56.68218°, 24.73467°	Transitional mire
			Kazu Mire	57.27769°, 24.82021°	Transitional mire
		Raised bog	Lielsala Mire	57.34391°, 22.31912°	Raised bog
			Ķemeri Mire	56.87991°, 23.53509°	Raised bog
Disturbed peatland	Drained, peat has not been extracted previously	Forest	Nelaubitis 1	56.68866°, 25.81881°	Middle-aged Norway spruce forest
			Nelaubitis 2	56.68880°, 25.81911°	Middle-aged Norway spruce forest
	Drained, peat have been extracted previously	Abandoned peat extraction site (bare peat)	Cena Mire	56.82469°, 23.98031°	Abandoned peat extraction site not covered with vegetation (bare peat)
			Medemas Mire	56.84537°, 24.10886°	Abandoned peat extraction site not covered with vegetation (bare peat)
		Abandoned peat extraction site with ground vegetation	Cena Mire	56.82370°, 23.97979°	Abandoned peat extraction site with ground vegetation
			Cepļa Mire	57.22008°, 26.47591°	Abandoned peat extraction site with ground vegetation
		Active peat extraction site (bare peat)	Kaigu Mire	56.75713°, 23.59221°	Active peat extraction site not covered with vegetation (bare peat)
			Ušuru Mire	57.03207°, 26.65576°	Active peat extraction site not covered with vegetation (bare peat)
		Forest	Viršu Mire	56.84503°, 24.10710°	Broadleaved (Silver birch) forest on peatland (12 years old)
			Cepļa Mire	57.21663°, 26.47955°	Coniferous (Norway spruce) forest on peatland, >20 years old
			Lielsala Mire	57.35506°, 22.32453°	Broadleaved (Silver birch) forest on peatland, >20 years old
			Kaigu Mire	56.74530°, 23.60078°	Coniferous (Norway spruce) forest on peatland, >20 years old
		Perennial grassland	Kašķu Mire	56.91085°, 23.58174°	Perennial grassland on peatland
			Lielsala Mire	57.35663°, 22.32327°	Perennial grassland on peatland
		Commercial berry plantation	Kaigu Mire	56.71175°, 23.60297°	Commercial blueberry plantation on peatland
			Kalna Mire	57.45605°, 26.45604°	Commercial cranberry plantation on peatland
			Kalna Mire	57.45563°, 26.45783°	Commercial blueberry plantation on peatland
			Ušuru Mire	57.03258°, 26.65852°	Commercial cranberry plantation on peatland

2.2. GHG Sampling and Measurements

During the measurement period (May–August 2019), soil GHG flux monitoring was conducted with the closed opaque manual chamber method [55]. At each research site, five chamber collars were evenly distributed with at least 3 m between individual collars. The collars were installed in approximately 5 cm of soil at least one month before the collection of the first GHG flux samples. Root damage was avoided as much as possible and ground vegetation and the litter layer, if present, were left intact during collar installation and field surveys, therefore, the monitored GHG flux represents the total soil emissions, including the heterotrophic respiration and autotrophic respiration of above- and below-ground vegetation enclosed in the soil collar and chamber during GHG sampling. Once per month, 4 soil flux samples were taken from chambers at each of the collar positions within 30 min of each other (10 min between each sampling) after positioning chambers on the collars. Due to potential diurnal patterns of soil GHG emissions [55] dynamic schedule of study site visits were applied to randomise gas sample collection time of the day [56]. The samples were collected in 100 mL vials at 0.3 mbar underpressure and transported to the laboratory (University of Tartu) to be tested with gas chromatography [57].

During GHG sampling, several environmental factors were determined: the groundwater (GW) level was measured manually inside a PVC pipe installed up to 140 cm deep in the soil at each research site; soil moisture and temperature by measurement probe inserted 5 cm into the soil and the air (ambient) temperature was taken with Comet data logger with temperature sensor.

2.3. Soil Sampling and Chemical Analysis

To avoid disturbing the soil inside the GHG chamber collars, it was sampled at two fixed depths (0–10 cm and 50 cm) on the outside opposite sides of each of the five collars at the research site. To better represent each research site, soil composite samples were made to represent two depths at the research site level. The 0 cm reference was at the top of the peat layer (the H horizon). Soil samples were taken using a 50-cm-long stainless-steel soil sample probe, sterilized instruments and plastic containers. Soil sampling was conducted in June–August 2019. Soil samples were transported to the LVS EN ISO 17025:2018 accredited laboratory at the Latvian State Forest Research Institute Silava and were prepared for analyses according to the LVS ISO 11464:2005 standard.

The THg content in the soil samples was determined with thermal decomposition, amalgamation and atomic absorption spectrophotometry (Milestone DMA—80 AC-N) according to the United States Environmental Protection Agency (US EPA 7473). The soil sample analysis results of THg < 1 µg kg^{-1} (n = 5) were replaced by half of the method limit of detection (0.5 µg kg^{-1}). The following parameters of general chemistry were determined: pH (KCl) according to the LVS EN ISO 10390:2022; organic C (OC, in g kg^{-1}), total N (TN, in g kg^{-1}) and total sulphur (TS, in mg kg^{-1}) content was determined with the elementary analysis method per the LVS ISO 10694:2006, LVS ISO 13878:1998 and ISO 15178:2000, respectively; the HNO_3-extractable phosphorus, potassium, calcium, magnesium and iron (respectively, P, K, Ca, Mg and Fe, in g kg^{-1}) content was determined with the inductively coupled plasma-optical emission spectrometry (ICP-OES) method and the electrical conductivity (conductivity, in µS cm^{-1}) was determined per the LVS ISO 11265:1994.

In addition, the OC/TN (C/N) ratio and OC/TS (C/S) ratios were calculated as proxies to characterize the decomposition of soil organic matter (SOM) [58,59]. To compare the Hg concentrations in soils and Hg storage, the relationships between THg and the major biophilic elements of the SOM (respectively, the THg/OC (Hg/C) ratio, the THg/TN (Hg/N) ratio and the THg/TS (Hg/S) ratio) were also calculated to overcome the effects of organic matter accumulation [51,52,58,60–64].

2.4. Vegetation Survey

A vegetation survey was conducted at all 22 research sites in summer (Table 1). At each study site, five circular sample plots were inventoried. The selected plots coincided with the edges of the installed chamber collars for GHG assessment. All vascular plant species, bryophytes and lichens were recorded, and the percentage coverages of each species were determined in the established plots. In total, 110 circular plots were described.

2.5. Statistical Analysis

A Wilcoxon rank-sum test with continuity correction was used to evaluate possible differences in THg concentrations, the values of the Hg/C, Hg/N and Hg/S ratios and the mean GHG emissions from the soil according to the pooled research soil in groups of management-induced disturbance and between soil depths (0–10 and 50 cm), with $p < 0.05$ considered significant. Correlations between THg concentrations, GHG emissions, the selected variables of soil general chemistry, environmental factors and vegetation cover were tested with Spearman's ρ, using a significance level of $p < 0.05$.

Soil chemical variables, environmental factors and vegetation cover variables (X) were used to explain the variance of THg concentrations and GHG emissions from soil (Y) via partial least squares (PLS) regressions. PLS regression is a useful multivariate method to address chemical variables that are linearly related to each other as the method is robust against intercorrelations among X variables. In PLS, X variables are ranked according to their relevance to explaining Y, commonly expressed as variables important for projection (VIP values). VIP values exceeding 1.0 are considered important X variables [65–67].

Statistical analyses (Wilcoxon rank sum test with continuity correction, Spearman's ρ and PLS) were performed with R [68]; the R package 'mdatools' was used for PLS. Figures 2 and 3 were prepared with the R package 'ggplot2', Figure 4 was prepared with the R packages 'corrplot' and 'Hmisc', Figures 5 and 6 were prepared with the R package 'ggplot2'.

A canonical correspondence analysis (CCA) was applied to assess the differences in species composition related to environmental variables. The species abundance (per cent coverage) data were used in the ordination as the main matrix and the environmental variables—the research site (abandoned peat extraction site, commercial berry plantation or active peat extraction site and coniferous forest, grassland, raised bog, broad-leaved forest or transitional mire), THg, CH_4, CO_2, N_2O, herbaceous cover, Sphagnum species coverage and total vegetation cover—as the second matrix. The CCA was carried out in PC-ORD 6 [69].

3. Results

3.1. Management-Induced Disturbance and Environmental Factors' Impact on the Soil THg Concentration

The spatial variation in soil THg concentrations at 0–10 cm deep across research sites was relatively high and ranged from <1 μg kg^{-1} in undisturbed pristine peatland (transitional mire) to 194.4 μg kg^{-1} in a research site disturbed by drainage that currently supports coniferous forest. At 50 cm deep, soil THg concentrations varied within a narrower range, from <1 μg kg^{-1} in both undisturbed and disturbed research sites to 75.8 μg kg^{-1} in undisturbed pristine peatland (raised bog). At the individual research site level, soil THg concentrations at 50 cm deep were mostly lower than at 0–10 cm deep, except at two disturbed research sites (an abandoned peat extraction site with ground vegetation and a commercial berry plantation) and one research site located in undisturbed pristine peatland (transitional mire). The difference between the soil THg concentrations at 0–10 cm and 50 cm deep (the concentration in the upper soil layer minus the concentration in the deeper soil layer), at each site, varied from −41.2 μg kg^{-1} in undisturbed pristine peatland (transitional mire) to 166.3 μg kg^{-1} in a research site disturbed by drainage and currently supporting a coniferous forest. When the mean THg concentrations in soil samples at 0–10 cm and 50 cm from all research sites were compared (Figure 2), statistically higher mean THg concentrations were found at 0–10 cm in disturbed research sites (67.6 ± 14.2 and 17.7 ± 3.7 μg kg^{-1}, respectively, $p = 0.004$); the differences in mean THg concentrations at 0–10 cm and 50 cm deep in undisturbed research sites were not statistically significant. Neither at a depth of 0–10 cm nor at 50 cm were statistically significant differences in mean THg concentrations between disturbed and undisturbed research sites found ($p = 0.902$ and $p = 0.313$, respectively).

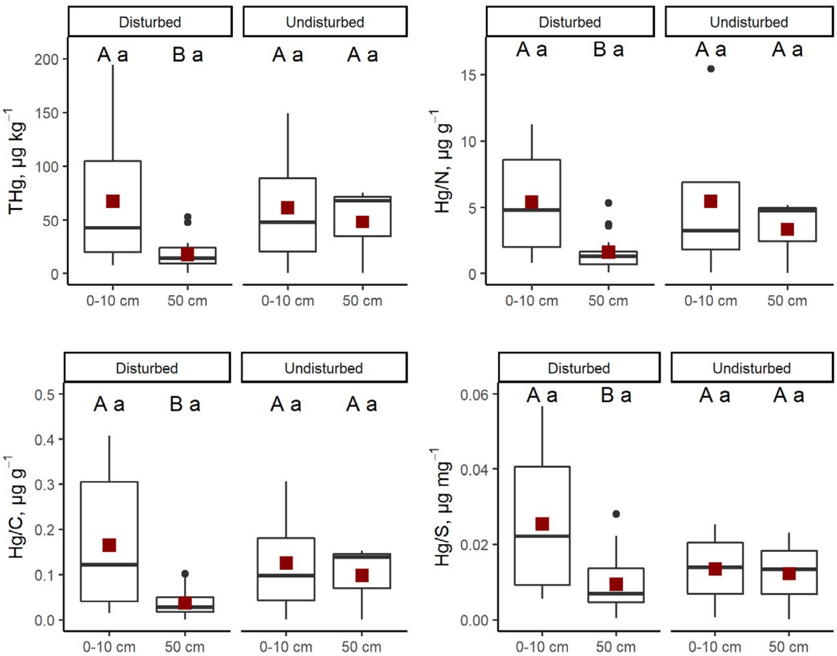

Figure 2. THg concentrations and relationships between Hg and the biophilic elements of soil organic matter (Hg/C, Hg/N and Hg/S ratios) in organic soil at 0–10 cm and 50 cm deep, grouped by management-induced disturbance. In the box plots, the median is shown by the bold line, the mean is shown by the dark red square, the box corresponds to the lower and upper quartiles, the whiskers show the minimal and maximal values (within 150% of the interquartile range from the median) and the black dots represent outliers of the datasets. Different uppercase letters show statistically significant differences ($p < 0.05$) between depths within the same group of management-induced disturbance; different lowercase letters show statistically significant differences ($p < 0.05$) between disturbed and undisturbed research sites within the same depth.

At a depth of 0–10 cm, the mean value of the Hg/C ratio at all research sites was 0.16 ± 0.03 μg Hg g^{-1} C (up to 0.41 μg Hg g^{-1} C). At 50 cm, the Hg/C values occupied a narrower range (up to 0.15 μg Hg g^{-1} C), and the mean value at all research sites was 0.05 ± 0.01 μg Hg g^{-1} C. The mean value of the Hg/N ratio at all research sites was 5.42 ± 0.91 μg Hg g^{-1} N (from 0.08–15.42 μg Hg g^{-1} N) at a depth of 0–10 cm and 1.86 ± 0.39 μg Hg g^{-1} N (0.05–5.36 μg Hg g^{-1} N) at 50 cm. The mean value of the Hg/S ratio was 0.023 ± 0.004 μg Hg mg^{-1} S (up to 0.057 μg Hg mg^{-1} S) at 0–10 cm deep and 0.010 ± 0.002 μg Hg mg^{-1} S (up to 0.028 μg Hg mg^{-1} S) at 50 cm.

At both analysed depths, the management-induced disturbance was not identified as a factor that introduced significant variation in the Hg/C, Hg/N and Hg/S ratios. However, as was found for THg concentrations, comparing the mean values of the Hg/C, Hg/N and Hg/S ratios at 0–10 cm and 50 cm deep (Figure 2) revealed statistically higher mean ratio values at 0–10 cm in disturbed research sites ($p = 0.002$, $p < 0.001$, and $p = 0.002$, respectively). The differences in the mean values of the Hg/C, Hg/N and Hg/S ratios between 0–10 cm and 50 cm deep in undisturbed research sites were not statistically significant.

The relationships of the Hg/C ratio to the C/N and C/S ratios in the soil at 0–10 cm reflect a logarithmic increase of the Hg/C ratio with the decay of SOM (Figure 3). The Hg/C and the C/N and C/S ratios displayed negative significant correlations at 0–10 cm (Figure 3) and 50 cm deep.

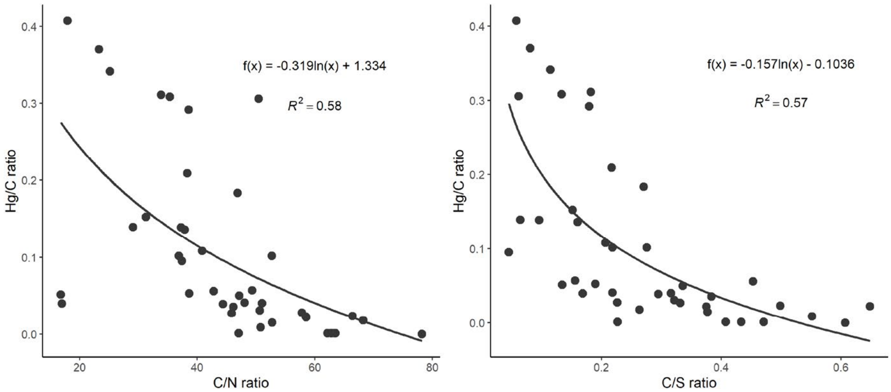

Figure 3. Relationships of the Hg/C ratio to the C/N and C/S ratios in the soil at 0–10 cm deep.

The general soil chemistry and GHG fluxes from the soil and vegetation cover of the peatlands were screened for relationships with the soil THg concentration (Figure 4). The soil THg concentration at 0–10 cm was positively correlated with TP (ρ = 0.66), TN (ρ = 0.65), TS (ρ = 0.65) and Ca concentrations (ρ = 0.43), but negatively correlated with the soil C/S ratio (ρ = −0.59), soil temperature during the measurement period (May–August) (ρ = −0.55) and the soil C/N ratio (ρ = −0.49) (Figure 4). Additionally, a PLS model revealed that the variation in the soil THg concentration at 0–10 cm between research sites was explained by soil chemistry parameters such as the TP, TN and TS concentrations and the soil C/S ratio (1.35 > VIP > 1.0), the soil temperature during the measurement period (VIP = 0.98) and the soil C/N ratio and Ca and Fe concentrations at 0–10 cm deep (VIP = 0.88, VIP = 0.77 and VIP = 0.65, respectively). The PLS model including these parameters had a goodness of fit (R^2) of 0.66 and a goodness of prediction (Q^2) of 0.57, indicating a moderate model. The variables that were negatively related to the THg concentration were the soil temperature during the measurement period and the C/S and C/N ratios.

The soil THg concentration at 50 cm deep was positively correlated with CH_4 emissions from the soil (ρ = 0.71) and the soil TS concentration at 50 cm (ρ = 0.59) but negatively correlated with the soil C/S ratio (ρ = −0.67) and the soil C/N ratio (ρ = −0.54).

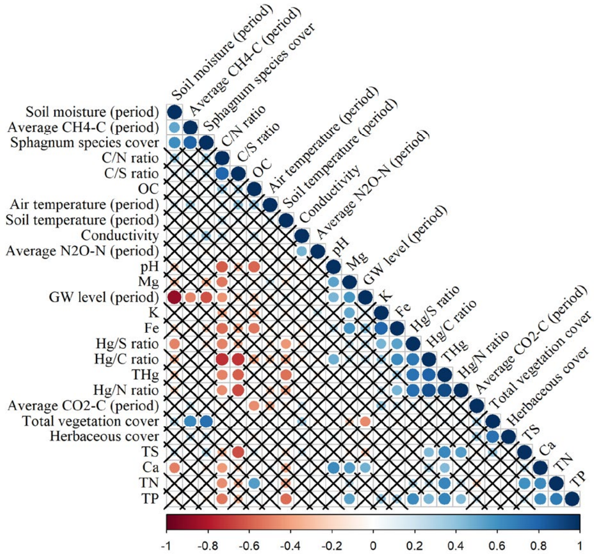

Figure 4. Spearman's correlations between the THg concentrations in soil at 0–10 cm deep, the mean GHG emissions from the soil during the measurement period (May–August 2019), the soil's general physico-chemical parameters at 0–10 cm and different environmental factors and vegetation cover. Positive correlations are displayed in blue and negative correlations in red. Colour intensity and the size of the circle are proportional to the correlation coefficients. In the right side of the correlogram, the legend colour shows the correlation coefficients and the corresponding colours. Correlations with $p > 0.05$ are considered as insignificant (crosses are added).

3.2. Management-Induced Disturbance and Environmental Factors' Impact on GHG Emissions from the Soil

The mean CO_2 emissions (sum of autotrophic and heterotrophic respiration) from research site soil during the measurement period (May–August) ranged from 20.1 $\pm$ 5.0 mg CO_2-C m^{-2} h^{-1} (abandoned peat extraction site, bare peat) to 104.6 $\pm$ 22.7 mg CO_2-C m^{-2} h^{-1} (research site disturbed by drainage and peat extraction, currently managed as grassland). The mean N_2O emissions from research site soil ranged from -0.97 µg N_2O-N m^{-2} h^{-1} (research site disturbed by drainage, currently coniferous forest) to 13.4 $\pm$ 11.6 µg N_2O-N m^{-2} h^{-1} (research site disturbed by drainage and peat extraction, currently managed as a commercial blueberry plantation). The highest spatial variation across research sites was found for mean CH_4 emissions—ranging from 30.8 $\pm$ 0.7 µg CH_4-C m^{-2} h^{-1} (research site disturbed by drainage and peat extraction, currently coniferous forest) to 3448.9 $\pm$ 1087.8 µg CH_4-C m^{-2} h^{-1} (undisturbed site, transitional mire).

Comparing the mean GHG emissions from the soil in undisturbed and disturbed research sites (Figure 5) revealed a statistically significant difference only for CH_4 emissions ($p < 0.001$). The higher CH_4 emissions of undisturbed research sites are related to soil moisture conditions. This is confirmed by the negative correlation between average CH_4 emissions and GW level ($\rho = -0.49$) and sequentially positive correlations between average CH_4 emissions and soil moisture ($\rho = 0.52$), Sphagnum species cover ($\rho = 0.81$) and total vegetation cover ($\rho = 0.65$) (Figure 4). A PLS model revealed that the variation in average CH_4 emissions between the research sites was explained by the average soil moisture, Sphagnum species cover and total vegetation cover (1.6 > VIP > 1.0). Although the PLS model including these parameters, as well as those with a 1.0 > VIP > 0.5 PLS model (average GW level, average air temperature, herbaceous cover, soil conductivity at 0–10 cm deep and average CO_2 emissions from the soil), had a goodness of fit (R^2) of 0.61, the goodness of prediction (Q^2) was 0.27, indicating a weak model.

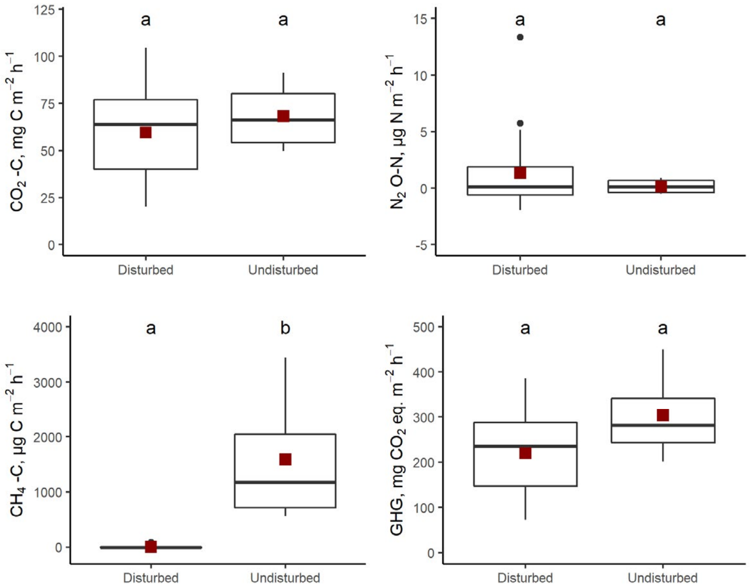

Figure 5. GHG emissions from organic soils during the measurement period (May–August 2019) in hemiboreal Latvia. In the boxplots, the median is shown by the bold line, the mean is shown by the black dot, the box corresponds to the lower and upper quartiles, whiskers show the minimal and maximal values (within 150% of the interquartile range from the median) and dots outside the box and whiskers represent outliers of the datasets. Different lowercase letters show statistically significant differences ($p < 0.05$) between disturbed and undisturbed research sites.

Soil CO_2 emissions were negatively correlated with soil OC content ($\rho = -0.47$), but soil N_2O emissions were positively correlated with soil electrical conductivity ($\rho = 0.46$) (Figure 4).

3.3. Vegetation Survey

In total, 103 species were recorded at the studied sites. The vascular flora was more diverse than the bryophytes and lichens. Altogether, 71 vascular plant species, 30 bryophyte species and two lichen species were recorded. Almost half of all determined bryophytes belonged to the *Sphagnum* genus (14 species). The undisturbed study sites were mostly covered by *Sphagnum* species, while the *Sphagnum* genus presented very low coverage at disturbed sites (Figure 6).

CCA ordination showed the relationships between species, research sites and environmental variables. The eigenvalues for axes 1 and 2 were 0.926 and 0.906, respectively. The variable THg was correlated with axis 1 (the Pearson and Kendall correlations were 0.763). In turn, the variables CH_4, *Sphagnum* species cover and Hg/C ratio were associated with axis 2 (the Pearson and Kendall correlations were −0.524, −0.618 and 0.645, respectively) (Figure 7).

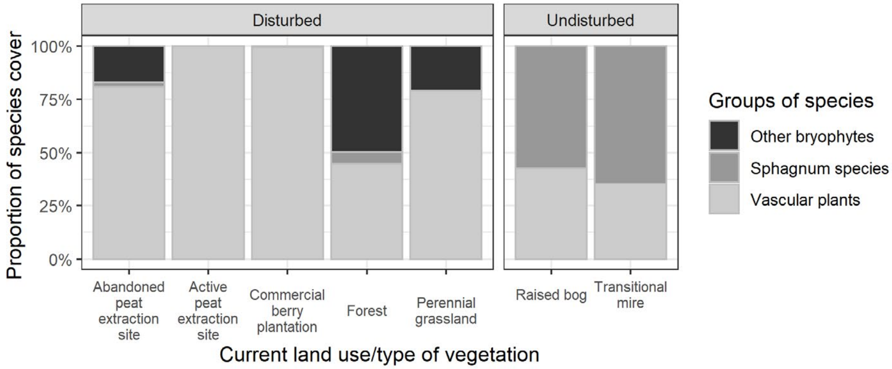

Figure 6. The proportion of species cover by different species groups and current land use or type of vegetation.

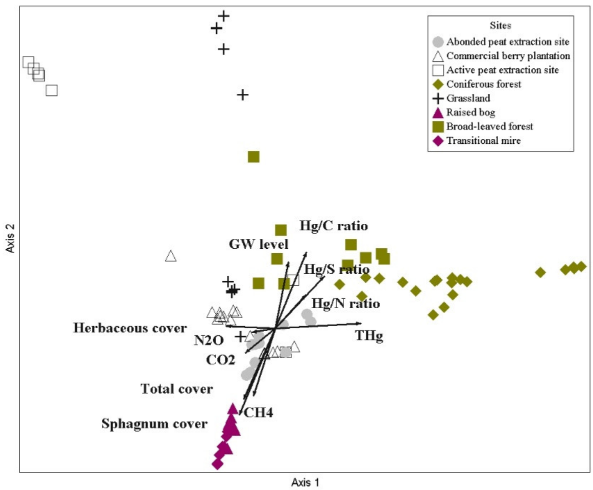

Figure 7. Canonical correspondence analysis (CCA) ordination of research site groups and environmental factors.

Two distinct species groups (clusters) were recognized in the CCA ordination. One of the species clusters was more related to study plots in forest sites (coniferous forest and broad-leafed forest), indicating higher mean values of THg per studied stand. The other species group was related to undisturbed sites—transitional mires and raised bogs. The results showed that undisturbed mires tended to have more CH_4 emissions, closer-to-soil-surface GW levels and greater *Sphagnum* species cover (Figure 7).

4. Discussion

4.1. Disturbance and Environmental Factors' Impact on Soil THg Concentrations

The THg concentrations in hemiboreal organic soils in peatlands ranged from <1 μg kg^{-1} to 194.4 μg kg^{-1} corresponding to concentrations found in uncontaminated soils [70–72]. Among peatland types, research sites disturbed by drainage and currently covered by coniferous forest had the highest average THg concentration in organic soil (103.0 ± 45.3 μg kg^{-1}, average from 0–10 cm and 50 cm deep).

In disturbed research sites, the mean soil THg concentration at 0–10 cm was statistically higher than at 50 cm, but in undisturbed research sites, no depth-related trends were observed, as in Giulio et al.'s study of North Carolina peatland [73]. Management-induced disturbance was found not to be a factor that introduced significant variation in THg concentration or Hg/C, Hg/N and Hg/S ratios. However, Hg cycling across peatland ecosystems (including Hg(II) methylation and demethylation processes) and exchange across the peat to surface water or atmosphere, including the uptake of Hg by vegetation and re-emission of gaseous elemental Hg, is complex [36,74–79]. From an international perspective, the average THg concentrations observed at the research sites (81.7 ± 17.8 μg kg^{-1} at 0–10 cm deep and 28.6 ± 6.9 μg kg^{-1} at 50 cm deep) in Latvia align with the concentrations reported for many other peatlands, although the overall variation in THg concentrations across and within peatlands is relatively large [46,58,80].

The strong relationship between Hg and SOM controls the transport and transformations of Hg in terrestrial ecosystems [46,81]. In natural systems, Hg binding to SOM occurs via thiol or other reduced S groups (mostly, Hg^{2+} predominates by bonding to two thiol groups or one thiol and either an N- or an O-containing group) [46,48,81]. Positive correlations between the soil THg and TS and TN concentrations at 0–10 cm depth were observed. This highlights the S and N functional groups as the key ligands for Hg retention in organic soils. Furthermore, S can enhance the net formation of MeHg by influencing both the activity of some types of Hg-methylating bacteria (as SO_4^{2-}) and the availability of Hg to methylating microorganisms (as S^{2-}), including those that do not use S-reduction in their metabolism [43,82–88]. In peatlands, where climate effects increase GW level fluctuation, previously reduced S can be re-oxidized or the opposite, resulting in further S legacy effects with potential consequences for MeHg production [39,84].

During SOM decomposition, C is lost from SOM more rapidly than N and S; thus, the C/N and C/S ratios reflect the process of SOM decomposition [58,63]. An increasing soil Hg/C ratio with decreasing SOM decomposition proxies (C/N and C/S ratios) was observed both at 0–10 cm and 50 cm deep (Hg/C and the C/N and C/S ratios correlated negatively and significantly). Similar trends were observed in forest soils in a recent study by Navrátil et al. [58] and Méndez-López et al. [52], where the Hg/C ratio trends were explained by the greater availability of Hg binding sites as organic matter decomposed. Soil N and S are usually positively correlated with organic matter [63] as they are in our study. Thus, the Hg/N and Hg/S ratios show similar trends to the Hg/C ratio.

There was a negative correlation between the soil temperature and THg concentration, a similar trend as the one found in a EU-level study along north-south gradient [72]. This is explained by enhanced Hg volatilization rate to the atmosphere with the temperature increase [89], a process that may have negative environmental consequences as global warming continues. MacSween et al. [90] predict that atmospheric warming by 1–2 °C may increase global Hg emissions by up to 43%.

Hg deposition could be affected by many factors, including differences in vegetation type and species composition. For instance, different vegetation types could affect the interception and retention of Hg differently [91,92]. Our results showed that the THg concentration varied between studied sites with different plant species compositions. The CCA ordination showed the tendency towards higher THg concentrations in forest-covered peatlands (broad-leafed and coniferous forests), while more open areas with higher *Sphagnum* species cover (undisturbed sites) had lower THg concentrations. In addition, the CCA ordination also indicated differences between forest types. A higher Hg concentration at 0–10 cm was more common in coniferous forests, but the Hg/C ratio was higher in broad-leafed forests. In addition, our results indirectly point towards the idea that the forest canopy could effectively collect Hg from the atmosphere through the tree leaves and, by litterfall and throughfall, mercury could be sequestered within the soil [79,92–95] and that coniferous trees have a higher capacity for Hg accumulation than deciduous trees [96].

In summary, this research shows the importance of vegetation as an influential factor for the deposition of Hg in the soil and of further study to better understand different Hg content in various ecosystems, especially as forested areas are one of the key sinks of Hg deposition in terrestrial ecosystems [94].

Vegetation tissue is not only important for supplying Hg but also to stimulate microbial activity, including methylation [75,97]. Non-vascular plants such as *Sphagnum* mosses (dominant in nutrient-poor bogs), tend to support acetogenesis and acetate accumulation. Vascular plants (dominant in richer fens), especially sedges, which can produce easily-degraded and high-quality C substrates via root exudation, support the accumulation of acetate, a low molecular weight organic substance used by bacteria as a C source to produce MeHg, for example, to a lesser degree [98].

4.2. Disturbance and Environmental Factors' Impact on Soil GHG Emissions

We compared GHG emissions (the sum of autotrophic and heterotrophic respiration, CH_4 and N_2O fluxes) from organic soil in undisturbed (pristine) and disturbed (managed) peatlands to examine the effect of management-induced disturbance and different environmental factors during the warmest season when, theoretically, the highest GHG emissions were expected as soil temperature is one of the main factors controlling GHG emissions [99]. Several studies have demonstrated that human-impacted peatlands (especially peatland-to-agriculture-converted sites) show significantly higher GHG emissions (mainly through N_2O and CO_2) than their natural counterparts [3,99,100], but our results revealed no significant differences in CO_2 and N_2O emissions between disturbed and undisturbed peatlands. Although slightly higher average CO_2 emissions were observed in undisturbed peatlands, a higher total variation in CO_2 emissions was observed in disturbed peatlands and, among peatland types, perennial grasslands showed the highest average CO_2-C (95.1 $\pm$ 9.5 mg CO_2-C m^{-2} h^{-1}) flux. High variation in CO_2 emissions monitored over a 2-yr period was observed among disturbed peatlands in Latvia also by previous study, furthermore, pristine peatlands tended to have even higher CO_2 emissions than some types of disturbed peatlands [34]. Similarly, a study in Scotland [101] revealed that CO_2 effluxes in lowland raised peat bog increased in the following order: undrained afforested < drained and afforested < pristine area of bog. Thus, our current results on CO_2 emissions are in line with previous findings in Latvia and elsewhere demonstrating that management effects are not always consistent in this regard.

Several studies have concluded that soil temperature, OC content in the soil, soil C/N ratio, soil bulk density and water table depth are the main environmental and soil chemistry factors explaining the amount and quality of respiring tissue and decomposing material, thus controlling CO_2 emissions within and between peatlands with different management history [18,99]. Our results show that variation in CO_2 emissions negatively correlates with OC content in organic soil. Thus, in peatlands where intensive peat mineralization occurs and OC content in soil is lower, higher CO_2 emissions are observed. No significant impact of GW level on CO_2 emissions was observed, likely because the GW level at research sites fluctuated widely both in disturbed and undisturbed research sites (from 13 to >130 cm from the soil surface with average 60 $\pm$ 3 cm and from 4 to 32 cm from the soil surface with average 16 $\pm$ 2 cm, respectively), during the study period. Thus, not only in disturbed but also in pristine peatlands GW level decreased below 10-cm layer where a major part of the new organic matter (including fine root litter) with the highest potential rate of decomposition is located, and, with reduction of water saturation and increase in aeration, the decomposition rate of this new organic matter increased [18]. In the region, in pristine peatlands, natural lowering of GW level below 20 cm from soil surface is usual especially in summer months and at the beginning of the autumn (e.g., [34]).

No significant impact of soil C/N ratio and only a weak impact of soil temperature measured at 0–5 cm deep on CO_2 emissions was observed during the study period. Weak correlation between soil temperature and CO_2 emissions may be explained by the limited temperature range in our study (covering only warm season), in combination with high variety of management practices with potentially different impact on emissions covered in research site group of disturbed peatlands. This results in highly variable vegetation composition and vegetation cover and subsequently in high variety of quality and quantity of vegetation litter which have significant impact on GHG emissions from soil [102].

Similar to CO_2 emissions, N_2O emissions do not show significant differences between disturbed and undisturbed peatlands, although slightly higher average N_2O emissions were observed in disturbed peatlands and, among peatland types, commercial berry plantations showed the highest average N_2O emissions (5.1 ± 3.0 μg N_2O-N m^{-2} h^{-1}). Several studies have concluded that N content, C/N ratio and soil temperature are the main factors controlling N_2O emissions [30,99]. No clear trends emerged. This could be explained by data obtained from peatlands with different current management and land-use history, potentially including fertilization and ploughing. The combination of these management methods may challenge the development of models to estimate N_2O emissions [18]. Our results show that variation in N_2O emissions positively correlates with soil conductivity, which is a good indicator of soluble salt concentrations in soil affecting the activity of soil microorganisms, which in turn influence such key soil processes as GHG emissions [103].

In contrast with CO_2 and N_2O emissions, CH_4 emissions from undisturbed peatlands were significantly higher than those from disturbed sites. Pristine transitional mire was the largest emitter of CH_4-C (2.1 ± 1.3 mg C m^{-2} h^{-1}) during the study period. The higher CH_4 emissions at undisturbed research sites are related to soil moisture conditions. Despite large fluctuations both at disturbed and undisturbed research sites, the mean GW level at undisturbed sites was still considerably higher. A lower water table directly reduces the production and increases the oxidation of CH_4 in the soil [18]. Our results indicate that GW level has a more significant impact on CH_4 emissions than on CO_2 emissions. Furthermore, a positive correlation between average CH_4 emissions and *Sphagnum* species cover was observed, although, in general, non-vascular plants such as *Sphagnum* mosses tend to inhibit terminal processes such as methanogenesis, while vascular plants, especially sedges, support increased methanogenesis by importing substrate to methanogenic microbes in anoxic soil layers and exporting CH_4 to the atmosphere past the methanotrophic microbes [98,104]. Conversely, bryophytes have been proven to predict CH_4 flux better than vascular plants, except for sedges. This is related to bryophytes' ability to better indicate the GW level long-term, thus reflecting zones of CH_4 production year-round [105].

Apart from contributing to the rather scarce data on GHG emissions from soils and THg concentrations in hemiboreal peatlands, our study also provides insight into differences between disturbed and undisturbed sites. In general, our results show that peatland management causes considerable changes in ecosystem processes, resulting in a high variation in environmental factors potentially affecting (directly and indirectly) GHG emissions from soil and THg concentration in peatland soils. Targeted ecosystem management to restore and enhance natural ecosystem functions is crucial to sustainable delivery of peatland ecosystem services. At the same time, the restoration efforts may simultaneously have contrasting effects on the cycling of biophilic elements of SOM (including C and N cycling) and contaminants, and the effects may differ in different biogeoclimatic regions. Not only current management decisions made on a local or regional scale, but also any broader policy aimed at promoting the restoration of a particular set of ecosystem functions should carefully consider all implications of the proposed measures. Complex, highly instrumented studies of ecosystem processes on a wider set of research sites where various parameters are assessed simultaneously and over a longer period of time will provide the much-needed basis for practical recommendations in peatland management.

5. Conclusions

Results revealed complex impacts of management-induced disturbance and environmental factors on the variation in THg concentrations and GHG emissions. The management-induced disturbance impact was mostly indirect, driving changes in environmental factors and vegetation cover. The most apparent impact of peatland disturbance was observed on CH_4 emissions, which were significantly higher in pristine peatlands.

Our results highlight the need for complex studies in managed peatlands, including a wider set of research sites and vegetation surveys, to clearly identify factors that may enhance Hg accumulation and increase GHG emissions as these sites harbour a high diversity of environmental variables and vegetation. As both Hg cycling and GHG emissions are largely microorganism-driven processes, microbial analysis should be included in further studies.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/land11091414/s1, Figures S1–S11: Visualizations of research sites.

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NITROUS OXIDE (N_2O) AND METHANE (CH_4) FLUXES FROM TREE STEMS IN BIRCH AND BLACK ALDER STANDS – A CASE STUDY IN FORESTS WITH DEEP PEAT SOILS

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Abstract. The aim of the study is to evaluate greenhouse gas (GHG) fluxes from stems in black alder (*Alnus glutinosa* (L.) Gaertn.) and birch (*Betula pendula* Roth) stands with drained and naturally wet nutrient rich peat soils, as well as to evaluate correlation between the GHG fluxes, soil temperature and groundwater level. The study was implemented in 8 forest stands – three black alder stands with nutrient rich peat soil (stand types according to national classification – *Dryopterioso-caricosa* and *Filipendulosa*) and 5 birch stands with peat soil (stand type *Oxalidosa turf. mel*. and *Dryopterioso-caricosa*). Measurement of GHG fluxes was continued for 12 months using Gasmet DX4040 FTIR analyser and removable non-transparent chambers of fixed volume and area. GHG fluxes were measured at 0.5, 1.0 and 1.5 m height on 3 trees in every stand. According to the study results the average CH_4 emissions from stem surface in birch stands are 6.9 ± 6.2 g CO_2 eq $m^{-2}\cdot yr^{-1}$ and in black alder stands 1.0 ± 3.2 g CO_2 eq $m^{-2}\cdot yr^{-1}$. Groundwater level significantly effects CH_4 emissions – if it remains above 15 cm during summer, the CH_4 emissions from stem increases to 84.0 ± 25.2 g CO_2 eq. $m^{-2}\cdot yr^{-1}$. Tree stems in drained peat soils are not a source of CH_4 emissions. According to the study results tree stems in peat soils are not producing N_2O emissions.

Keywords: GHG, emissions, tree stems, alder, birch.

Introduction

Organic soils are the largest source of GHG emissions in Latvia contributing to more than 6,1 mill. tons $CO_2\cdot yr^{-1}$ according to the national GHG inventory [1]. However, only part of the sources of GHG emissions in organic soils can be reported using country specific emission factors [2]. To calculate emissions from organic soils in Latvia, so far country specific methods [3-5] and default methods provided by the IPCC guidelines [6ж 7] have been used. The review of the methods proposed by the IPCC guidelines points to large diversity of scientific approaches applied in the referred studies leading to large uncertainty of the elaborated emission factors [8]. This review also highlights importance of elaboration of the country specific methodological approaches for evaluation of GHG fluxes from organic soils.

The recent studies in neighbouring countries prove that trees can be a significant source of methane (CH_4) emissions, especially in areas with seasonally fluctuating or continuously high groundwater level. Increase of CH_4 emissions during seasonal floods and periodic increase of the groundwater level can contribute to more than 70% of the net CH_4 emissions in forests with water saturated soils [10]. This and earlier studies [10ж 11] studies have significantly clarified the processes affecting GHG fluxes in organic soils and pointed to underestimated sources of GHG emissions – pristine, naturally wet organic soils and tree stems. Comprehensive studies are necessary to prove the effect of certain climate change mitigation measures, e.g., seasonal adjustment of groundwater level in deciduous tree stands and use of selective harvesting (openings and bends of limited area) instead of regenerative clear-felling. Limited and controversial knowledge about GHG fluxes in organic soils in combination with high uncertainty hampers implementation of climate change mitigation measures aimed at reduction of the largest source of GHG emissions in Latvia.

The urgent need to improve knowledge base required to eliminate GHG emissions in organic soils is also determined by the Regulation (EC) No. 2018/841, recently published proposal for amendment of the regulation [12; 13] and the European Commission communication document No. COM(2018)773. According to the amendment to the regulation No. 2018/841 the neutrality target in LULUCF sector is set in 2030, requiring reduction of GHG emissions by at least 4 mill. tons CO_2 eq$\cdot yr^{-1}$ [14].

Accounting of GHG emissions and CO_2 removals in LULUCF sector in Latvia recently has been significantly improved, because of LIFE REstore project [15-17] and other studies. However, CH_4 fluxes from tree stems are not yet addressed resulting in potential underestimation of GHG emissions in forests with organic soils. This is limiting the ability to forecast the climate effect of different forest management scenarios.

To address the most urgent needs of the climate policy in Latvia's LULUCF sector this project is aimed at evaluation CH_4 and N_2O fluxes from the tree stems in birch and black alder stands and to evaluate the effect of the groundwater level and other factors on the CH_4 and N_2O emissions. The study results are unique at European level and are applicable in countries with similar climatic conditions.

Materials and methods

GHG measurements were implemented in eight forest stands – tree black alder stands with nutrient rich peat soil (stand types according to national classification – *Dryopterioso-caricosa* and *Filipendulosa*) and five birch stands with peat soil (stand type *Oxalidosa turf. mel* and *Dryopterioso-caricosa*). Additionally, stem fluxes from birch were measured in one of the black alder stands. Information on stands including location is provided in Table 1. The study was implemented from November 2020 till October 2021, 12 months. Frequency of sampling – once per two weeks between April and October and once per month during winter months (in total 20 measurement campaigns).

Table 1

Stand characteristics in measurement plots

Dominant species	Stand ID	Age	Height, m	Diameter, cm	Basal area, $m^2 \cdot ha^{-1}$	Density, $trees \cdot ha^{-1}$	Location, WGS84	
							X	Y
Birch	031-99-9	20	15	14	21	1180	57.3218	26.0641
Birch	502-457-2	30	17	12	20	498	56.6873	25.0482
Birch	504-408-3	59	21	27	16	462	56.6942	24.5836
Black alder	508-45-11	23	11	9	17	1890	56.6596	24.1421
Birch	012-186-1	60	16	18	25	1243	57.2906	25.9987
Birch	501-20-15	70	22	22	11	289	56.9289	24.9666
Black alder	501-20-17	53	24	22	12	265	56.9280	56.9280
Black alder	505-84-3	72	26	29	31	584	56.5737	56.5737

GHG fluxes were measured at 0.5, 1.0 and 1.5 m height on three trees in every stand, excluding the stand where two plots – for measurement of the fluxes from birch and black alder were installed. Height and diameter of the measured trees are provided in Table 2.

Measurement of GHG fluxes was done using Gasmet DX4040 FTIR analyser and removable non-transparent chambers of fixed volume and area. Different chambers (Table 3) were used depending on the diameter of trees. Before the measurement the area of the bark surface, where the chamber is attached to the stem surface, was treated with silicone to avoid air exchange, when the chamber is installed.

Measurement continued for 30 minutes per tree, simultaneously at all heights. Manual multiplexers were used to switch between different chambers. Content of gases was determined after installation of the chamber and after 8, 10, 18, 20, 28 and 30 minutes. If different intervals are used, it is noted out during the measurement and later considered in the calculation. In parallel to the flux measurements, the groundwater level, soil and air temperature were recorded.

Table 2

Dimensions of the measured trees

Dominant species	Stand ID	Diameter of trees at 1.3 m height, cm			Height of trees, cm		
		tree 1	tree 2	tree 3	tree 1	tree 2	tree 3
Birch	031-99-9	20.5	16.4	11.5	18.5	17.8	16.0
Birch	502-457-2	20.3	14.7	12.7	21.4	18.2	19.4
Birch	504-408-3	28.6	25.9	19.5	22.7	22.4	18.2
Birch	508-45-11	14.5	10.6	9.2	12.6	12.2	12.0
Birch	012-186-1	20.5	18.0	12.9	22.4	21.3	20.2
Birch	501-20-15	29.9	21.0	14.0	25.4	23.6	20.8
Black alder	508-45-11	11.3	11.1	8.7	11.4	11.4	9.7
Black alder	501-20-17	24.3	19.3	14.2	23.7	21.3	19.8
Black alder	505-84-3	36.9	23.9	21.2	30.2	27.1	26.7

Table 3

Dimensions of the measurement chambers

Chamber ID	Height, cm	Width, cm	Thickness, cm	Volume, m³	Area, m²
1	20.1	25.0	2.2	0.00111	0.05025
2	20.5	42.0	2.3	0.00198	0.08610
3	20.0	56.5	2.5	0.00283	0.11300
4	19.0	73.0	2.8	0.00388	0.13870

R^2 of the linear regression of the CO_2 concentration changes is used to ensure that outliers are excluded from the flux calculation. Only data series with $R^2 > 0.95$ are used in the calculation. GHG fluxes were calculated using the following equation [15]:

$$CO_2 - C\left[\mu gC \cdot m^{-2} \cdot h^{-1}\right] = \frac{M\left[g \cdot mol^{-1}\right] * P[Pa] * V\left[m^3\right] * \delta v\left[ppm \cdot h^{-1}\right] * f_1 * f_2 * f_3}{R\left[m^3 \cdot Pa \cdot K^{-1} \cdot mol^{-1}\right] * T[K] * t[h] * A\left[m^2\right]},$$

where P – air pressure in the chamber, assumed constant 101300 Pa);
V – chamber volume, m^3 (Table 3);
δv – slope of regression representing gas concentration changes per hour;
R – universal gas constant (8.3143 $m^3 \cdot Pa \cdot K^{-1} \cdot mol^{-1}$);
T – soil temperature, K;
t – measurement time, hours;
M – molar mass of measured gases, 16.04 CH_4, $g \cdot mol^{-1}$;44.01 N_2O, $g \cdot mol^{-1}$;
A – chamber surface area, m^2 (Table 3);
f_1, f_2 and f_3 – recalculation coefficients (Table 4).

Table 4

Coefficients for calculation of GHG fluxes

Gas	f_1	f_2	f_3
CH_4	0.75	1.00	1.00
N_2O	0.64	1.00	1.00

Emissions were extrapolated to an area by calculation of the stem surface of an average tree, assuming that it is cone and by multiplication the average surface area with the number of trees per ha. Branches are not considered in the estimation due to lack of published information on the surface area of crown and a ratio between the GHG fluxes from stem and from branches.

Results and discussion

CH_4 emissions were observed in birch stands during summer months. In autumn, spring and winter months no CH_4 emissions were observed (Fig. 1). In black alder stands only one occurrence of significant CH_4 emissions was found in spring. The main reason for the difference was higher groundwater level in several birch stands. No N_2O emissions were observed during most of the time, birch stem surface is acting as net sink of N_2O removals; however, the effect is negligible.

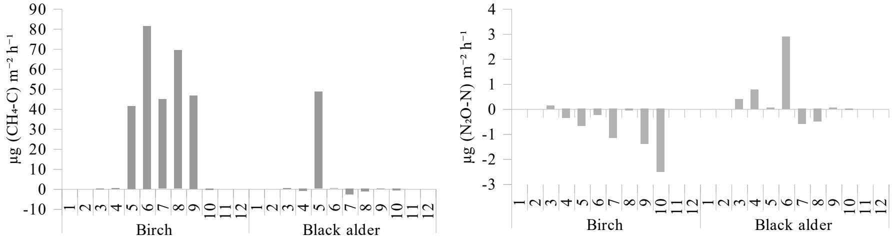

Fig. 1. **Results of measurements – average monthly fluxes**

Comparison of GHG fluxes and the groundwater level demonstrates significant correlation with CH_4 emissions in birch stands and no correlation in black alder stands (Fig. 2). It was also found that only in two birch stands the groundwater level increased above 15 cm during the vegetation season. As soon as the groundwater level drops below 15 cm, no CH_4 emissions from the stem surface appear; however, when the groundwater level increases, particularly in spring and summer, CH_4 emissions increase. No correlation was found with N_2O emissions and the groundwater level.

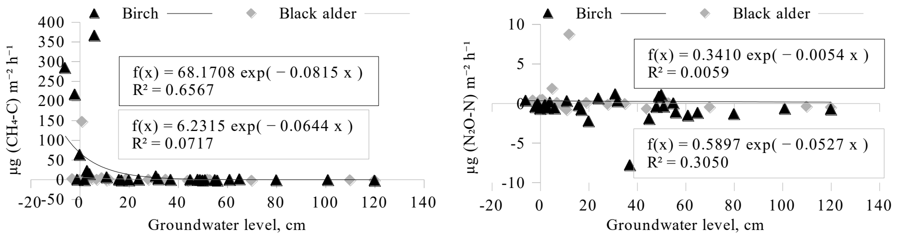

Fig. 2. **Correlation between groundwater level and GHG fluxes**

Increase of soil temperature is also increasing CH_4 emissions; however, only in case of high groundwater level (Fig. 3). No correlation was found between temperature and N_2O emissions from stems.

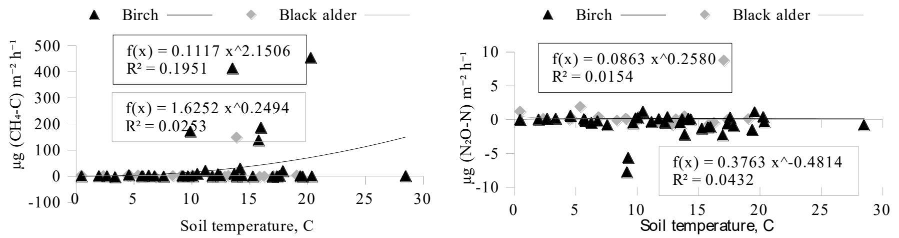

Fig. 3. **Correlation between soil temperature and GHG fluxes**

According to the study results the emissions seem to be more relevant to the groundwater level than the species, since the emissions are determined only if the groundwater level remains above 15 cm. If the groundwater level is high during most of the vegetation season or the area is flooded, the CH_4 emissions from stem increase to 325 ± 81 kg CO_2 eq. $ha^{-1}\cdot yr^{-1}$ (84.0 ± 25.2 g CO_2 eq. $m^{-2}\cdot yr^{-1}$). These results point out the importance of regulation of the water regime to eliminate hotspots of CH_4 emissions in forest lands with organic soils. According to earlier studies [15], CH_4 emissions in flooded areas equal to 100.6 CH_4, kg CH_4-C $ha^{-1}\cdot yr^{-1}$ (3.3 tons CO_2 eq $ha^{-1}\cdot yr^{-1}$). According to this study results stem fluxes in average conditions are negligible; however, high groundwater level or increase of the groundwater level during the vegetation period significantly increases CH_4 emissions. In one of the birch stands stem fluxes of CH_4 reached 10% of the total CH_4 emissions from soil and stem surface, if the soil CH_4 emission factor applied in the national GHG inventory is used to estimate CH_4 emissions from soil. The study does not approve findings by other authors, e.g. [15] that increase of the groundwater level in alder stands increases N_2O emissions. This may be associated with different periods of the increase of the groundwater level, in our study it was high in alder stands in spring, till June. Significant increase of CH4 emissions from the stem surface due to increase of the groundwater level is reported by several authors, e.g. [10; 18]. According to these authors changes are correlating with soil fluxes – reduction of CO_2 emissions and increase of CH_4 emissions from soil.

Conclusions

1. The research proves the results of earlier studies that the deciduous tree in organic soils can be a significant source of CH_4 emissions, while no significant N_2O emissions are detected.
2. Tree stem surface becomes a source of CH_4 emissions only in areas, where the groundwater level is above 15 cm, and the emissions rapidly grow if the groundwater level is higher.
3. CH_4 emissions are correlating also with temperature; however, the correlation is weak and CH_4 emissions only increase in case of high groundwater level, therefore both factors – groundwater level and temperature – should be used in projections of CH_4 emissions from the tree stem surface.
4. Significant improvements of activity data (dynamic maps of groundwater level) are necessary to estimate CH_4 emissions from tree stems at a national or regional scale.

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Author contributions

Methodology, A. L. and A.B..; data analysis, R.A.; writing – review and editing, A. L., A. B., R. A. All authors have read and agreed to the published version of the manuscript.

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CARBON DIOXIDE (CO_2) EMISSIONS FROM NATURALLY WET AND DRAINED NUTRIENT-RICH ORGANIC FORESTS SOILS

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Abstract Implementation of climate change mitigation measures in forestry has a key role to successfully fulfil the climate change policy goals of Land use, land use change and forest sector set by the Paris Agreement to fully offset total GHG emissions in the country by CO_2 removals in 2050. GHG emissions from organic soils in forest land have significant impact on total emissions of Latvia, however, high emissions also indicate the potential of climate change mitigation measures. This study aims to evaluate CO_2 emissions from drained and naturally wet nutrient-rich forest soils to improve knowledge of forest management practice impact on GHG emissions. The study is conducted in 21 drained (*Myrtillosa turf.mel.* and *Oxalidosa turf. mel.*) and 10 naturally wet (*Dryopterioso–caricosa and Filipendulosa*) forest sites with nutrient-rich organic soils for 12 consecutive months. Soil total CO_2 emissions were measured by closed manual non-transparent chamber method. The groundwater level, soil and air temperature were measured to evaluate factors affecting CO_2 emission. Empirical data collected within the scope of the study showed high correlation ($r = 0.85$) between CO_2 emissions and temperature, however, the groundwater level depth had no considerable impact on emissions. Total soil CO_2 emissions from drained nutrient-rich organic soils ranged from 5.44 t ± 0.1 tC·ha^{-1}·yr^{-1} in black alder stands to 9.76 ± 2.47 tC·ha^{-1}·yr^{-1} in clearcut areas (average 7.35 ± 0.89 tC·ha·yr^{-1}), while CO_2 emissions from forest sites with naturally wet soil ranged from 5.73 ± 2.23 tC·ha^{-1}·yr^{-1} in spruce stands to 10.41 ± 4.33 tC·ha^{-1}·yr^{-1} in clearcut areas (average 7.02 ± 0.96 tC·ha^{-1}·yr^{-1}). The study results demonstrate that drainage does not have significant effect on CO_2 emissions.

Keywords: organic soil, naturally wet, drained, CO_2 emissions

Introduction

Organic soil is one of the largest carbon (C) storages of terrestrial ecosystems globally [1] and also in Latvia [2]. Depending on the land use and management practices organic soil can act as C sink or source [3]. Share of organic soils is 19% of total area of Latvia [4]. According to the national forest site type classification system [5] and information provided by the national forest inventory (NFI) the area of organic soils in forest land is 723 kha of which 53% are drained.

According to the Intergovernmental panel on Climate change (IPCC) guidelines [6] for National GHG inventories human induced GHG emissions shall be estimated – regarding organic forest soils only GHG emissions from drained and rewetted organic soils are reported in Latvia, respectively. IPCC guidelines divide organic soils as nutrient-poor and nutrient-rich, however, if the IPCC default methodology is applied, the most of forest organic soils in Latvia can be considered as nutrient-rich, since they are receiving nutrients with groundwater and precipitation. The IPCC default emission factor (2.6 t CO_2-C·ha^{-1}·yr^{-1}) for calculation of CO_2 emissions from drained organic forest soils [7] is replaced by the national emission factor 0.52 t CO_2-C·ha^{-1}·yr^{-1} developed as a result of multiple studies evaluating long term C stock changes after drainage [2; 8; 9]. The national emission factor characterises emissions from organic soils in forest site types *Callunosa turf. mel.*, *Vacciniosa turf. mel.* and *Myrtillosa turf.mel.* with nutrient-poor to moderate-rich soils [10], yet it is applied to all drained organic forest soils in the national GHG inventory. While for rewetted organic soils the IPCC default emission factor 0.5 t CO_2-C ha^{-1}·yr^{-1} is used [7]. According to this approach total estimated human induced CO_2 emissions from organic forest soils in forest lands were almost 800 kt CO_2 or 7% of total GHG emissions in Latvia in 2020.

Although it is not mandatory to report GHG emissions from naturally wet organic forest soils, information on such emissions is necessary to elaborate and implement knowledge-based climate change mitigation measures in forest management to work towards climate neutrality policy goals set by the Paris Agreement, as well as to provide scientifically substantiated assessment of the effect of drainage and rewetting of forest soils. This study aims to work towards better understanding of differences between the net CO_2 emissions from drained and naturally wet nutrient-rich organic forest soils.

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Materials and methods

The study was conducted in 31 forest sites in central Latvia with nutrient-rich (the most fertile site types for drained and naturally wet soils) organic soils [5]. One sample plot (500 m^2) was established in each of the selected forest stands: 10 sample plots in naturally wet (*Dryopterioso–caricosa and Filipendulosa*) and 21 sample plots in drained (*Myrtillosa turf.mel*. and *Oxalidosa turf. mel*.) sites (Table 1). Dominant tree species of Norway spruce, silver birch, black alder, as well as 1 year old clearcuts of deciduous, mixed stands. To check the forest site conformity to the specified site type the peat depth was determined (threshold value of at least 20 cm in drained and 30 cm in naturally wet soils by 5 measurement replicates. Additionally, ground floor vegetation was characterized to select areas representing plant communities typical for certain site types. The centre of the sample plots was at least 20 m from the stand border.

Table 1

Study site characteristics

Parameter	Value	Naturally wet forest sites				Drained forest sites			
		Norway spruce	Silver birch	Black alder	Clearcut	Norway spruce	Silver birch	Black alder	Clearcut
Number of study sites	number	1	3	5	1	12	3	2	4
Age of dominant tree species, years	Average	67	56	43	-	55	39	40	-
	range (min...max)	-	21-77	10-80	-	14-86	18-60	26-53	-
Growing stock, m^3·ha^{-1}	average	446	225	170	-	269	135	189	-
	range (min...max)	-	78-365	35-325	-	7-521	38-210	123-254	-
Peat layer, cm	average		41	59	47	81	43	65	90
	range (min...max)	-	31-52	23-99	-	37-99	25-75	60-70	63-99

During the study period from October of 2019 till June of 2021 soil CO_2 emissions were monitored for 12 consecutive months by the closed manual non-transparent chamber method [11], when the mean air temperature was 9.2 ± 0.8 ºC (min 8.0 ± 0.7, max 31.4 ± 0.1) and annual precipitation 668 ± 136 mm (ranged from 472 mm to 860 mm) according to 5 meteorological stations in a range of up to 30 km from the sample plots. 5 collars were installed in every plot at least 1 month prior the first CO_2 emission measurement. Sides of the collars reached approximately 5 cm depth. Roots were not trenched and ground vegetation as well as the litter layer were left intact, therefore CO_2 emissions measured include soil heterotrophic and both above- and belowground ground vegetation autotrophic respiration (soil total emissions).

Sample plots were surveyed once per month by taking 4 gas samples from each chamber position on each collar installed. Gas samples were collected with the interval of 10 minutes: 0; 10; 20 and 30 minutes after carefully positioning chambers on the collars. The samples collected in underpressurized 100 mL glass vials were transported to the laboratory to be analysed by a gas chromatograph equipped with an electron capture detector [12]. Simultaneously with gas sampling the air and soil temperature at 5 cm depth, as well as the ground water level in the groundwater level monitoring wells (140 cm long PVC pipe) installed at time of establishment of the sample plots were measured.

Soil total CO_2 emissions are estimated by using the slope acquired form the linear regression curve representing CO_2 concentration changes in the chamber during the measurement period of 30 minutes. For quality assurance purpose only slopes with $R^2 > 0.7$ were used for further analysis. The ideal gas equation is used for calculation of soil total CO_2 emissions:

$$CO_2 = \frac{MPVslope}{RTtA}, \quad (1)$$

where CO_2 – soil total CO_2 emissions, µg CO_2 $m^2 \cdot h^{-1}$;
M – molar mass of CO_2, $g \cdot mol^{-1}$;
R – universal gas constant, 8.314 $m^3 \cdot Pa \cdot K^{-1} \cdot mol^{-1}$;
P – assumption of air pressure inside the chamber, 101 300, Pa;
T – air temperature, K;
V – chamber volume, 0.063 m^3;
t – time, 1 h;
slope – CO_2 concentration changes in time, $ppm \cdot h^{-1}$;
A – collar area, 0.1995 m^2.

All soil total CO_2 emission measurement results in the paper are expressed in unit of $tC \cdot ha^{-1} \cdot yr^{-1}$, indicated uncertainty is the confidence interval. Data compliance to normal distribution is checked by Shapiro-Wilk test and differences of mean values – by Mann-Whitney test. Significance level $\alpha = 0.05$ is applied in statistical analysis.

Results and discussion

According to the data acquired in the study monthly average total CO_2 emissions from soil in naturally wet and drained sites are not significantly different ($p = 0.25$) and ranged from 2.39 to 15.81 $tC \cdot ha^{-1} \cdot yr^{-1}$ in February and June, accordingly (Fig. 1).

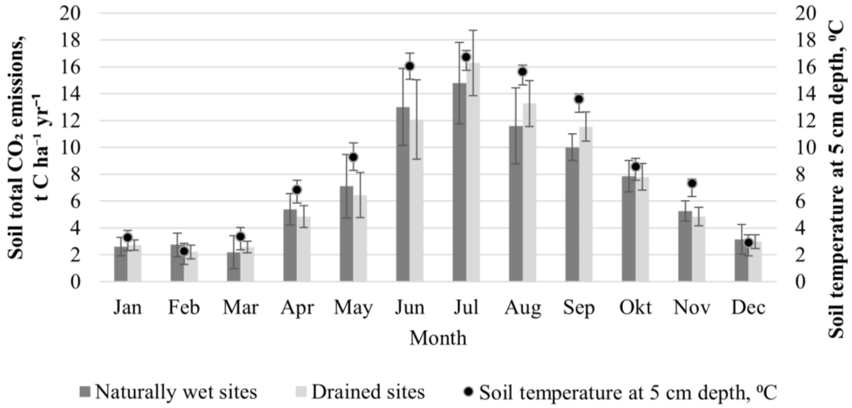

Fig. 1. **Variation of monthly soil total CO_2 emissions**

The groundwater level has low correlation with CO_2 emissions ($r = -0.30$). Variations of soil CO_2 emissions can be explained with changes of the soil temperature. Relationship of CO_2 emissions and soil temperature at 5 cm depth is characterized by exponential regression (Fig. 2). While the air and soil temperature have significant correlation (r = 0.89) characterised by linear equation:

$$t_{soil} = 0.64 t_{air} + 1.96 \quad (2)$$

where t_{soil} – soil temperature at 5 cm depth, ºC
t_{air} – air temperature, ºC.

Total annual CO_2 emissions from soil range from 5.44 ± 0.10 to 9.76 ± 2.47 $tC \cdot ha^{-1} \cdot yr^{-1}$ in drained black alder dominated stands and clearcuts and from 5.81 ± 2.23 to 10.55 ± 4.33 $tC \cdot ha^{-1} \cdot yr^{-1}$ in naturally wet Norway spruce stands and clearcuts, accordingly (Fig. 3). The impact of drainage conditions (naturally wet or drained soil) on the total CO_2 emissions from soil with different dominant tree species and difference of the mean annual emission between the dominant tree species is not significant ($p < 0.05$).

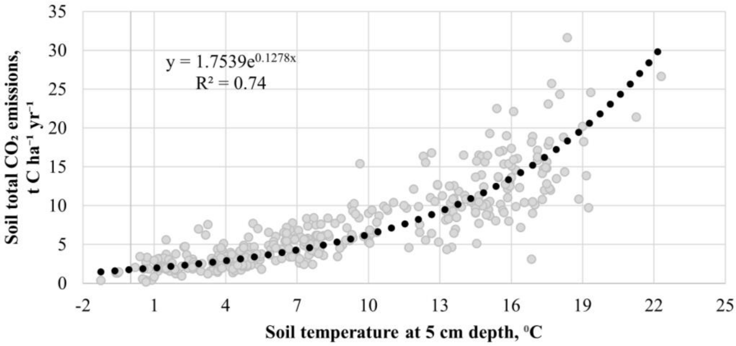

Fig. 2. **Relationship between soil total CO_2 emission and soil temperature**

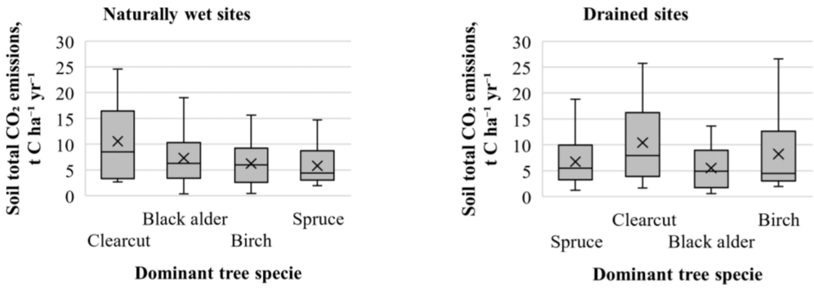

Fig. 3. **Intra annual variation of soil total CO_2 emissions in forest stands with different dominant tree species and soil drainage status**

Consequently, statistically significant differences between mean total CO_2 emissions from soil in different forest site types ($p > 0.05$) as well as in drained (7.35 ± 0.89 tC·ha^{-1}·yr^{-1}) and naturally wet (7.02 ± 0.96 tC·ha^{-1}·yr^{-1}) study sites ($p = 0.34$) were not found (Fig. 4). Intra annual total CO_2 emissions from soil in the study sites with tree cover ranged from 0.38 to 31.66 tC·ha^{-1}·yr^{-1}, while in clearcuts – from 0.17 to 25.74 tC·ha^{-1}·yr^{-1}. It was found that the CO_2 emissions above 22.13 tC·ha^{-1}·yr^{-1} are statistical outliers as indicated in Fig. 4 and differences between annual mean CO_2 emissions in forest stands (6.84 ± 0.56 tC·ha^{-1}·yr^{-1}) and clearcuts (10.08 ± 1.96 tC·ha^{-1}·yr^{-1}) are statistically significant ($p = 0.002$).

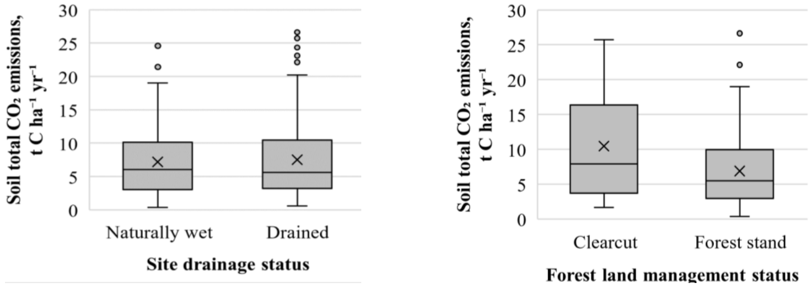

Fig. 4. **Intra annual variation of soil total CO_2 emissions in different forest site types and by drainage status and tree cover**

It is important to note that total reported CO_2 emissions from soil are gross soil emissions and include both soil heterotrophic and autotrophic respiration and do not consider soil C input by above-and belowground litter. It is reported in similar ecosystems studied that total CO_2 emissions from soil can be recalculated to heterotrophic respiration by the factor 0.5 [13-15]. It is estimated according to the NFI data on tree species and age distribution in Latvia that the weighted mean annual carbon input with above ground and belowground litter in drained organic soil is 0.27 ± 0.01 $tC \cdot ha^{-1} \cdot yr^{-1}$ and 0.65 ± 0.01 $tC \cdot ha^{-1} \cdot yr^{-1}$ in silver birch and Norway spruce stands, accordingly; while weighted average C annual input by fine roots is 1.43 ± 0.07 $tC \cdot ha^{-1} \cdot yr^{-1}$ in Norway spruce dominated stands and 1.70 ± 0.07 $tC \cdot ha^{-1} \cdot yr^{-1}$ in silver birch stands; and annual carbon input by tree foliar litter is 2.0 $tC \cdot ha^{-1} \cdot yr^{-1}$ and 1.86 $tC \cdot ha^{-1} \cdot yr^{-1}$ in silver birch and Norway spruce dominated stands with basal area of 20 m^2 ha^{-1} [16]. By combining the above mentioned data on soil CO_2 emissions an C input and estimating combined uncertainty, annual net soil CO_2 emissions are $-0.55 \pm 0{,}29$ $tC \cdot ha^{-1} \cdot yr^{-1}$ in silver birch stands and -0.52 ± 0.29 $tC \cdot ha^{-1} \cdot yr^{-1}$ in Norway spruce stands. To improve the net soil CO_2 emission estimate, additional national data on soil carbon input stratified by dominated tree species, soil fertility and drainage status as well as the proportion of heterotrophic and autotrophic respiration are necessary.

Conclusions

1. The study results show no significant impact of the forest site type, dominant tree species or drainage status on annual mean total CO_2 emissions from nutrient-rich organic soils (sum of soil heterotrophic and ground vegetation autotrophic respiration).
2. Differences between annual mean total CO_2 emissions from nutrient-rich organic soils in forest stands ($6.84 \pm 0~.56$ $tC \cdot ha^{-1} \cdot yr^{-1}$) and clearcut areas ($10.08 \pm 1.96$ $tC \cdot ha^{-1} \cdot yr^{-1}$) are statistically significant.
3. Combining the study results on the CO_2 emissions from nutrient-rich organic soils with the estimates from earlier studies on the soil C input in forest sites with drained organic soils, the calculated net CO_2 emissions from the soil in the studied areas in silver birch stands are -0.55 ± 0.29 $tC \cdot ha^{-1} \cdot yr^{-1}$ and -0.52 ± 0.29 $tC \cdot ha^{-1} \cdot yr^{-1}$ in Norway spruce stands; respectively, they are net sinks of CO_2 removals.

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Author contributions

Writing – original draft preparation, A.B.; field work, G.S.; writing – review and editing, I.L.; laboratory work – analysis of collected samples and data management, D.P.

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N_2O AND CH_4 EMISSIONS FROM NATURALLY WET AND DRAINED NUTRIENT-RICH ORGANIC FOREST SOILS

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According to general knowledge rewetting of drained organic soils is a measure that can reduce net greenhouse gas emissions from ecosystem, however there is lack of evidence that approves such an assumption in hemiboreal forests. The aim of the study was to quantify N_2O and CH_4 flux from nutrient-rich organic soils in naturally wet (NWS) and drained (DS) hemiboreal forest sites in Latvia.
In central Latvia, 26 NWS (*Dryopterioso–caricosa* and *Filipendulosa*) and DS (*Oxalidosa turf. mel.*) were selected to evaluate annual N_2O and CH_4 soil flux by manual chamber method. Gas sampling was performed once a month in five replicates in every sampling plot for period of one year covering all seasons from October of 2019 till November of 2020. During gas sampling soil temperature and groundwater level were measured. In addition, soil and groundwater was sampled and tested.
Study results show that soil CH_4 flux has strong correlation with groundwater level and weak correlation with soil temperature in booth DS and NWS. Moderate correlation between soil temperature and N_2O flux were found in DS, however in rest of the study sites significant impact of soil temperature and groundwater level on N_2O flux was not found. Estimated annual average soil CH_4 flux is average -3.5±1.0 kg C-CH_4 ha^{-1} yr^{-1} in DS and average 100.6±101.0 kg C-CH_4 ha^{-1} yr^{-1} in NWS. While estimated annual average soil N_2O flux is average 1.1±0.4 kg N-N_2O ha^{-1} yr^{-1} in DS and 2.6±0.9 kg N-N_2O ha^{-1} yr^{-1} in NWS.

Keywords: drained organic soil, naturally wet organic soil, CH_4 flux, N_2O flux

INTRODUCTION

Paris agreement signed by 195 parties worldwide, in enhancing the implementation of the United Nations Framework Convention on Climate Change (Convention), aims to hold the increase in the global average temperature to well below 2 °C above pre-industrial levels (United Nations..., 2015). Despite the efforts dedicated for reaching climate mitigation goals greenhouse gas (GHG), including nitrous oxide (N_2O) and methane (CH_4), concentration in atmosphere continues to increase. According to data of National Oceanic and Atmospheric Administration and Advanced Global Atmospheric Gas experiment, since the Convention took into force on 1994 till 2018, total GHG concentration in atmosphere has been consistently increasing by 17.2 %, from 389.6 to 456.8 ppm CO_2 eq. (Prinn et al., 2021). It is estimated if GHG concentration in atmosphere persists between 430 and 480 ppm CO_2 eq. in 2100, probability of exceeding atmospheric temperature increase threshold of 1.5 °C is 49 to 86 % (Clareke et al., 2014). During period from 1994 till 2016, CH_4 and N_2O emissions in atmosphere have increased by 6 % from 1742 to 1842 ppb and from 311 to 329 pbb accordingly (Prinn et al., 2021) and continues to increase. Although GHG emissions, including emissions of land use, land use change and forestry (LULUCF), from European Union have been reduced by 27 % and reduction of N_2O and CH_4 is as high as 37 % since 1990 till 2018, N_2O and CH_4 emissions still constituted 18 % of total GHG emissions (in CO2 eq.) in 2018 (Mandl Nicole (EEA) et al., 2020). Neither N_2O and CH_4 emissions are a key source of LULUCF in EU level, however these emission from drained organic soils are a key source of LULUCF sector in national GHG inventory of Latvia (Latvia's National..., 2021; Mandl, Pinterits, 2020). Total area of forest organic soils in Latvia is 696.5 kha or 10.8 % of total state area, furthermore 54.8 % of organic forest soils are drained. CH_4 and N_2O emissions from drained and rewetted organic soils in forest lands accounted for 7.3 % of total national GHG emissions in CO_2 equivalents in 2019 (Latvia's National..., 2021).

Climate change mitigation targets set at global, European Union as well as at national levels has increased scientific focus on ecosystem GHG emission studies. Furthermore, Regulation of the European Parliament and of Council on the inclusion of greenhouse gas emissions and removals from LULUCF into the 2030 climate and energy framework promotes role of LULUCF sector in achieving climate change mitigation goals by setting a binding commitment to ensure that accounted emissions from land use are entirely compensated by CO_2 removals in LULUCF sector. Regulation aims to fully offset the country's total GHG emissions by CO_2 removals in the LULUCF sector in the second half of the 21st century. Furthermore, Proposal for a Regulation of the European Parliament and of the Council amending Regulation (EU) 2018/841 on the inclusion of GHG emissions and removals from LULUCF in the 2030 climate and energy framework aims to set a target of GHG removal of the LULUCF sector in 2030 thus making the sector even more crucial in reaching overall EU climate targets.

In the national GHG inventory of 2019 Latvia used default Tier 1 CH_4 (2.5 kg CH_4 ha^{-1} yr^{-1}) and N_2O (2.8 kg N_2O-N ha^{-1} yr^{-1}) emission factors (EF) from 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories (2013 Wetlands Supplement). To improve accuracy of Latvia's national GHG inventory and to support policy makers this study aims to elaborate national CH_4 and N_2O emissions factors for drained and naturally wet nutrient-rich organic soils.

RESEARCH METHODS

The study was conducted in 31 forest sites with nutrient-rich drained and naturally wet organic soils located in central Latvia from October of 2019 till November of 2021 (**Klaida! Nerastas nuorodos šaltinis.**). Annual average air temperature within the study period according to 5 closest meteorological stations within range of 30 km from atleast 1 study site was 9.2±0.8 ºC (min 8.0±0.7, max 31.4±0.1), while annual precipitation ranged from 472 mm to 860 mm (average 668±136 mm).

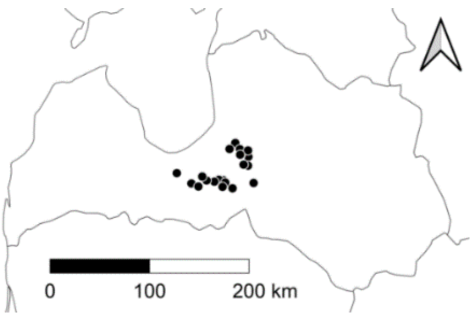

Figure 1. Location of study sites in Latvia

Selection of study sites

Primary study sites selection was based on site soil moisture regime and fertility characteristics according to the national forest site type classification system (Bušs, 1981). For further evaluation from 4 soil fertility classes forest stands characterized as compliant to 2 most fertile forest stands classes with drained (*Myrtillosa turf.mel.* and *Oxalidosa turf. mel.)* and naturally wet (*Dryopterioso–caricosa and Filipendulosa*) organic soils were selected. Sample plots were established in naturally wet and drained sites with peat layer atleast 30 cm and 20 cm accordingly (checked at atleast 5 places within sample plots). During the study period soil GHG monitoring was conducted for 12 consecutive months in each of the study sites. In each of the study site 1 round (500 m^2) sample plot was established atleast 20 m from forest stand or clearcut border. Soil GHG fluxes measurements were done by closed opaque manual chamber method (Pavelka et al., 2018). 5 chamber collars were installed evenly within sample plot with distance between individual collars at least 3 m. Collars were installed in soil depth aproximately 5 cm at least one month prior to first GHG measurements. Root damages were avoided as far as possible and ground vegetation was left intact during collar installation and field surveys. Sample plots were visited once per month and 4 soil flux samples were taken from chambers in each of collar positions within 30 minutes (10 minutes between each sampling) after positioning chamber on collar. Samples were collected in 100 mL vials with 0.3 mbar underpressure and transported to the laboratory to be tested by gas chromatograph. During gas sampling soil temperature at 5 cm depth as well as groundwater level was measured, in addition groundwater samples were collected from groundwater level monitoring wells for further tests in laboratory. For site fertility characterisation soil samples were collected from each sample plot in depth up to 80 cm (within step of 10 cm) (Cools and De Vos, 2016).

GHG flux samples were analysed in University of Tartu by gas chromatograph (Loftfield et al., 1997). Physio-chemical analysis of soil and water samples were done in Laboratory of Forest Environment of Latvian State Forest Research Institute "Silava". The soil samples were prepared for analyses according to the LVS ISO 11464 (2005) standard. Chemical parameters were determined to organic soil milled till fine powder and fine earth fraction (D < 2 mm) of mineral soil (prepared according to LVS ISO 11277) according to standard methods (Table 1). Organic carbon concentration (g kg^{-1}) in soil was calculated as the difference between total carbon concentration and inorganic carbon (carbonate) concentration. Water samples analysed by photometry and ion chromatography were filtered through 0.45 µm and 0.2 µm filters accordingly.

GHG flux calculation

GHG flux is calculated using slope of linear regression that represents hourly GHG concentration changes in chamber. Acquired slope data was discarded if R^2<0.7 except cases when difference between maximum and minimums concentration in chamber was less then gas chromatograph method uncertainty. Acquired slope information was further expressed as GHG flux from area of soil:

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

where *flux* – soil GHG flux, µg GHG m^2 h^{-1};
M – molar mass of GHG, g mol^{-1};
R – universal gas constant, m^3 Pa K^{-1} mol^{-1};
P – assumption of air pressure inside the chamber, 101 300 Pa;
T – air temperature, K;
V – chamber volume, 0.063 m^3;
t – time period between first and last GHG flux sampling, 0.5 h;
slope – slope of the hourly GHG concentration changes inside of chamber;
A – collar area, 0.1995 m^2.

Table 1. Standard methods utilised for soil and groundwater sample analysis

Parameter	Unit	Method principle	Standard method
Soil samples			
Bulk density	kg m^{-3}	Gravimetry	LVS ISO 11272:2017
Total carbon	g kg^{-1}	Elementary analysis (dry combustion)	LVS ISO 10694:2006
Total nitrogen	g kg^{-1}	Elementary analysis (dry combustion)	LVS ISO 13878:1998
$CaCO_3$	g kg^{-1}	Volumetry	ISO 10693
HNO_3 extractable K, Ca, Mg and P	g kg^{-1}	ICP-OES	LVS EN ISO 11885:2009)
Groundwater samples			
pH	log unit	Potentiometry	LVS ISO 10523:2012
Conductivity (EC)	µS cm^{-1}	Conductometry	LVS EN 27888:1993
Total nitrogen (N)	mg L^{-1}	Catalytic oxidation	LVS EN 12260:2004
Nitrates (NO_3^-), phosphates (PO_4^{3-})	mg L^{-1}	Ion chromatography	ISO 10304-1:2007
Ammonium ion (NH_4^+)	mg L^{-1}	Photometry	LVS ISO 7150-1:1984

Statistical analysis

Data statistical analysis was carried out using RStudio (Rstudio Team, 2019). The compliance of the data distribution with the normal distribution was checked using the Kalmogorov-Smirnov test. Statistical differences of GHG fluxes between forest site groups with drained and naturally wet soils were evaluated by Wilcoxon signed-rank test. Correlation between GHG flux and affecting factors were determined by Pearson and Spearman correlation. Data uncertainty within this paper is expressed as confidence interval, significance level α=0.05 is applied.

RESEARCH RESULTS

Characteristics of study sites

Mean peat layer in study sites ranged from 25 cm to at least 100 cm (average 75±7 cm) and 23 cm to at least 100 cm (average 54±12cm) in DS and NWS respectively. Study site topsoil (upper 20 cm layer) characteristics are summarised in

Table 2.

Table 2. Study site characteristics

Parameter	Value	Naturally wet forest sites: Norway spruce	Silver birch	Black alder	Clearcut	Drained forest sites: Norway spruce	Silver birch	Black alder	Clearcut
Number of study sites	number	1	3	5	1	12	3	2	4
Forest stand characteristics									
Age of dominant tree species, years	average	67	56	43	-	55	39	40	-
	range (min...max)	-	21-77	10-80	-	14-86	18-60	26-53	-
Growing stock, m^3 ha^{-1}	average	446	225	170	-	269	135	189	-
	range (min...max)	-	78-365	35-325	-	7-521	38-210	123-254	-
Peat layer, cm	average		41	59	47	81	43	65	90
	range (min...max)	-	31-52	23-99	-	37-99	25-75	60-70	63-99
Topsoil (upper 20 cm layer) characteristics									
C_{org}, g kg^{-1}	average ±SE	490	463±26	344±96	447	483±37	316±97	430±53	546±17
N_{tot}, g kg^{-1}	average ±SE	32	25±4	19±5	28	23±8	23±2	27±4	27±8
P, g kg^{-1}	average ±SE	1.9	1.2±0.6	1.7±0	3.8	1.5±0.3	2.1±0.6	3.2±0.7	1.3±0.1
K, g kg^{-1}	average ±SE	19	21±4	18±2	16	21±1	14±0.5	16±1	15±1
Ca, mg kg^{-1}	average ±SE	0.3	0.4±0.02	0.5	0.6±0.1	0.3±0.03	0.7±0.3	1.0±4	0.6±0.01
Mg, g kg^{-1}	average ±SE	18	10±6	14±4	42	16±2	24±8	32±8	12±3

During the study period of 1 year depth of groundwater level in both drained and naturally wet forest sites ranged from atleast 140 cm to 0 cm. Mean distance from topsoil to groundwater level was 55±2 cm and 35±3 cm at drained and naturally wet forest sites respectively. Monthly mean groundwater level was by 18±2 cm deeper in drained forest sites (Figure 2).

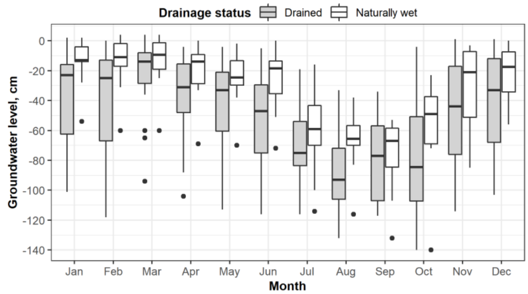

Figure 2. Monthly groundwater depth variation in study sites. In the boxplots, the median is shown by the bold line, the box corresponds to the lower and upper quartiles, whiskers show the minimal and maximal values (within 150% of the interquartile range from the median) and black dots represent outliers of the datasets.

Soil GHG flux and affecting factors

Study results shows that soil CH_4 flux is a subject of high uncertainty. Difference of estimated soil CH_4 flux within same survey of single study site reaches 2 and 4 orders of magnitude in DS and NWS accordingly, thereby spatial variability of soil CH_4 flux is considerable higher in NWS. During the study period estimated annual average soil CH_4 flux in DS ranged from -5.5±1.0 kg C-CH_4 ha^{-1} yr^{-1} in Norway spruce stands to 6.8±16.6 C-CH_4 ha^{-1} yr^{-1} in Black alder stands (average -3.5±1.0 kg C-CH_4 ha^{-1} yr^{-1}), while in NWS estimated soil CH_4 flux ranges from -3.7±2.8 kg C-CH_4 ha^{-1} yr^{-1} in Silver birch stands to 199.8±393.2 kg C-CH_4 ha^{-1} yr^{-1} in Black alder stands (average 100.6±101.0 kg C-CH_4 ha^{-1} yr^{-1}). Study results indicate that Black alder forest stands tend to have considerably higher average soil CH_4 flux compared to other tree species dominated forest stands included in this study, however also uncertainty of estimated annual soil CH_4 flux results for Black alder stands is considerable higher (Table 3). Pattern of exceedingly high emissions were found in 10 % of NWS.

Table 3. Annual soil CH_4 flux (kg C-CH_4 ha^{-1} yr^{-1}) in study sites

Dominant tree specie	Drained forest sites	Naturally wet forest sites
Silver birch	-1.7±2.0	-3.7±2.8
Norway spruce	-5.5±1.0	-2.4±1.2
Clearcut	-4.7±1.0	6.9±6.2
Black alder	6.8±16.6	199.8±393.2
Black alder (hotspot excl.)	-	-0.9±0.4
Black alder (hotspot)	-	10036.7±834.4
Average	-3.47±0.94	100.6±101.0

Estimated average soil CH_4 flux of Black alder stands ranges from -1.7±1.0 kg C-CH_4 ha^{-1} yr^{-1} to 15.5±12.7 kg C-CH_4 ha^{-1} yr^{-1} in DS (2 study sites) and from -1.9±1.1 kg C-CH_4 ha^{-1} yr^{-1} to 1036.7±834.4 kg C-CH_4 ha^{-1} yr^{-1} in NWS (5 study sites), furthermore if soil CH_4 flux hotspot site is excluded, average CH_4 flux from rest of 4 NWS

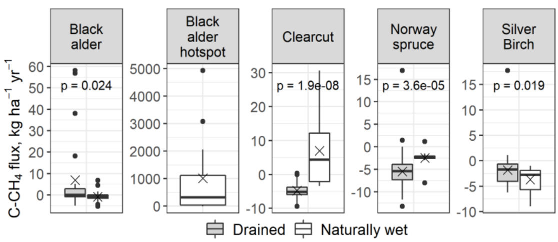

Figure 3. Intra-annual soil CH_4 flux variation. In the boxplots, the median is shown by the bold line, the mean is shown by “x”, the box corresponds to the lower and upper quartiles, whiskers show the minimal and maximal values (within 150% of the interquartile range from the median) and black dots represent outliers of the datasets.

study sites ranges from -1.9±1.1 kg C-CH_4 ha^{-1} yr^{-1} to -0.2±0.7 kg C-CH_4 ha^{-1} yr^{-1} (Figure 3**Klaida! Nerastas nuorodos šaltinis.Klaida! Nerastas nuorodos šaltinis.**).

Acquired soil CH_4 flux data has weak correlation with soil temperature and groundwater chemical analysis result data but has strong nonlinear correlation with groundwater level data in both DS and NWS, however it was not possible to elaborate model with good fit to raw empirical data set due to high proportion of CH_4 flux data outliers with considerably high concentrations. If outliers are excluded relationship between groundwater level and soil CH_4 flux is characterised by exponential regression (Figure 4).

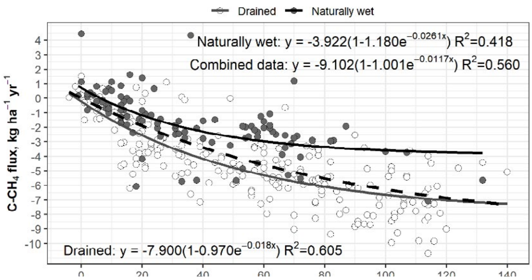

Figure 4. Relationship between groundwater level depth and soil CH_4 flux

These results indicate that during majority of measurements soil has not been a source of CH_4 emissions in booth DS and NWS, however as groundwater raised CH_4 removals decreased till gradually turned into CH_4 emissions as groundwater level reached topsoil and soil were saturated by water respectively. Similar observations are made if also statistical outliers are included in data evaluation. Regardless of drainage status soils become a source of CH_4 emissions when groundwater depth decreased below 20 to 30 cm. If whole dataset is considered average soil CH_4 flux from DS and NWS is significantly different in all groundwater depth ranges, except in depth between (0 to 9 cm) (p=0.27) (Table 4). Furthermore, if the one sample plot mentioned above with excessively high soil CH_4 flux at NWS is excluded from evaluation, average flux differences remain significant ($p<0.05$) in all groundwater depths except from 0-9 cm (p=0.95) and in conditions when GHG flux sampling ring is flooded (p=0.90).

Table 4. Average soil CH_4 flux by different groundwater level depths

Groundwater level, cm	Drained forest sites		Naturally wet forest sites					
	kg C-CH_4 ha y^{-1}	n	kg C-CH_4 ha y^{-1}	n	kg C-CH_4 ha y^{-1}	n	kg C-CH_4 ha y^{-1}	n
	Total data		Total data		Without hotspot site		Hotspot site	
Flooded	1.6±0.9	45	448.1±869.9	37	12.1±11.9	14	1025±1184.7	23
0-9	5.2±3.2	107	366.1±409.3	104	2.3±3.7	87	2233.5±2377.6	17
10-19	0.4±3.3	123	20.7±22.5	104	0.3±1.7	99	510.2±302.5	5
20-29	-2.8±0.4	105	-1.9±1	60	-1.9±1	60	-	0
30-39	-3.8±0.5	90	-2.7±1.1	60	-2.7±1.1	55	-2.1±1.6	5
40-49	-2.3±2.3	65	-2.4±0.6	25	-2.2±0.7	20	-3.3±1	5
50-59	-5±0.6	80	-2.1±1.2	65	-2.1±1.2	65	-	0
60-69	-5.1±0.5	105	-2.6±0.5	60	-2.6±0.5	55	-2.6±0.9	5
70-79	-5.6±0.5	115	-2.7±1.5	35	-2.7±1.5	35	-	0
80-89	-6.4±0.6	60	-3.9±1.2	20	-3.9±1.2	20	-	0
90-99	-7±0.6	70	-	0	-	0	-	0
100-119	-7.2±0.5	175	-5.6±1.1	20	-5.6±1.1	20	-	0
120-140	-5.8±1	20	-7.3±1.7	10	-7.3±1.7	10	-	0

According to the study results average annual soil N_2O flux in DS (1.1±0.4 kg N-N_2O ha^{-1} yr^{-1}) and NWS (2.6±0.9 kg N-N_2O ha^{-1} yr^{-1}) differ significantly (p=0.01). Average annual soil N_2O flux in DS ranged from 0.6±0.6 to 1.5±1.3 kg N-N_2O ha^{-1} yr^{-1} in Black alder dominated stands and clearcuts accordingly (Table 5). While in NWS highest average soil N_2O flux where found in Black alder dominated stands (3.3±4.0 N-N_2O ha^{-1} yr^{-1}) and lowest flux – in clearcut sample plot (0±0.1 kg N-N_2O ha^{-1} yr^{-1}). Furthermore, in case of Black alder dominated stands (p=0.001) and clearcuts ($p<0.05$) difference between DS and NWS soil N_2O flux is significant. According to data acquired, soil temperature had moderate (r = 0.48) impact on soil N_2O flux in DS only, while groundwater level had weak impact on N_2O flux in neither DS and NWS. From groundwater quality parameters monitored NO_3^- and N as well as Ca and Mg concentration had the most notable impact on soil N_2O flux. NO_3^- and N concentration had moderate linear correlation in DS (r = 0.54 and 0.52 accordingly) and weak linear correlation in NWS (r = 0.42 and 0.32 accordingly). While Ca and Mg concentration had

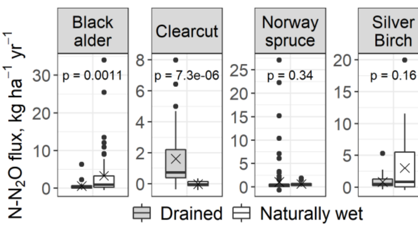

Figure 5. Intra-annual soil N_2O flux variation

weak nonlinear correlation in DS (r = 0.44 and 0.42 accordingly) and moderate correlation in NWS (r = 0.57 and 0.62 accordingly). Regarding pH and EC, weak linear correlation was found in NWS only (r = 0.42 and 0.43 accordingly).

Table 5. Annual soil N_2O flux (kg $N\text{-}N_2O$ ha^{-1} yr^{-1}) in study sites

Dominant tree specie	Drained forest sites	Naturally wet forest sites
Silver birch	0.9±0.6	2.7±3.1
Norway spruce	1.0±0.9	0.6±0.3
Clearcut	1.5±1.3	0±0.1
Black alder	0.6±0.6	3.3±4.0
Average	1.1±0.4	2.6±0.9

CONCLUSIONS AND DISCUSSION

Study results show that groundwater level depth threshold found for nutrient-rich organic forest soils to become a source of CH_4 emissions around 20 to 30 cm complies with assumption of 2013 Wetlands Supplement guidelines regarding drainage class classification – threshold of groundwater level depth of 30 cm to distinguish between shallow or deep drained soils (IPCC, 2014). Estimated average soil CH_4 flux in NWS monitored in this study (100.6±101.0 kg C-CH_4 ha^{-1} yr^{-1}) is similar but with considerably less uncertainty if compared to default EF for CH_4 from rewetted nutrient-rich organic soils in boreal climate zone (0 to 493 kg C-CH_4 ha^{-1} yr^{-1}, average 137 kg C-CH_4 ha^{-1} yr^{-1}) and considerably lower compared to EF for CH_4 from nutrient-rich organic soils in temperate climate zone (0 to 856 kg C-CH_4 ha^{-1} yr^{-1}, average 216 kg C-CH_4 ha^{-1} yr^{-1}) provided by 2013 Wetlands Supplement indicating that. Lower uncertainty is achieved also for calculated annual average soil CH_4 flux in DS (-3.47±0.94 kg C-CH_4 ha^{-1} yr^{-1}) compared to default EF for drained organic soils in temperate (-0.6 to 5.7 kg C-CH_4 ha^{-1} yr^{-1}, average 2.5 kg C-CH_4 ha^{-1} yr^{-1}) and drained nutrient-rich organic soil boreal (-1.6 to 5.5 C-CH_4 ha^{-1} yr^{-1}, average 2.0 C-CH_4 ha^{-1} yr^{-1}) climate zones. Estimated annual soil N_2O flux in both DS (1.1±0.4 kg N-N_2O ha^{-1} yr^{-1}) and NWS (2.6±0.9 kg N-N_2O ha^{-1} yr^{-1}) is within uncertainty of default N_2O EF for drained organic soils in temperate climate zone (-0.57 to 6.1 kg N-N_2O ha^{-1} yr^{-1}, average 2.8 kg N-N_2O ha^{-1} yr^{-1}) and EF for nutrient-rich drained organic soils in boreal climate zone (1.9 to 4.5 kg N-N_2O ha^{-1} yr^{-1}, average 3.2 kg N-N_2O ha^{-1} yr^{-1}).

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Evaluation of Soil Organic Layers Thickness and Soil Organic Carbon Stock in Hemiboreal Forests in Latvia

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Abstract: In the forest land of many European countries, including hemiboreal Latvia, organic soils are considered to be large sources of greenhouse gas (GHG) emissions. At the same time, growing efforts are expected in the near future to decrease emissions from the Land Use, Land Use Change and Forestry sector, including lands with organic soils to achieve enhanced contributions to the emissions and removals balance target set by the Paris Agreement. This paper aims to describe the distribution of organic soil layer thickness in forest land based on national forest inventory data and to evaluate soil organic carbon stock in Latvian forests classified as land with organic soil. The average thickness of the forest floor (organic material consisting of undecomposed or partially decomposed litter, O horizon) was greatest in coniferous forests with wet mineral soil, and decreased with increasing soil fertility. However, forest stand characteristics, including basal area and age, were weak predictors of O horizon thickness. In forests with organic soil, a lower proportion of soil organic matter layer (H horizon) in the top 70 cm soil layer, but a higher soil organic carbon stock both in the 0–30 cm layer and in the 0–100 cm layer was found in drained organic soils than in wet organic soils. Furthermore, the distribution of the soil H horizon thickness across different forest site types highlighted the potential overestimation of area of drained organic soils in Latvian forest land reported within the National GHG Inventory.

Keywords: hemiboreal forests; litter layer; organic soils; organic carbon stock

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1. Introduction

The carbon (C) stock in the world's forests including soil, live biomass, deadwood, and litter is estimated to be 861 ± 66 Gt C [1]. Globally, almost half of the total organic carbon (OC) in forest ecosystems is stored in the forest floor and in soils down to 1 m depth [1,2]. De Vos et al. (2015) estimated that forests in the European Union store ~3.7 Gt C in forest floors and ~22 Gt C in soils down to 1 m depth [3]. In general, soil organic carbon (SOC) stock reflects the equilibrium between inputs of organic matter produced mainly by overstory trees and understory vegetation to soils and the loss of C through decomposition, biotic respiration, leaching and erosion of soil organic matter [2]. As SOC stored and cycled in forests is a considerable share of the global C stock [1,4], even negligible changes in the SOC stock induced, for instance, by land management or climate change could have large impacts on the atmospheric carbon dioxide (CO_2) concentration and thereby accelerate global warming [5–7].

Although organic soils, especially in drained areas, are large sources of greenhouse gas (GHG) emissions in forest land of many European countries [8], forests are expected to increase CO_2 removals and to decrease GHG emissions [3,4,9] to achieve implementation of the climate change mitigation goals, such as those set by the Paris Agreement [10] and formulated in long-term low GHG emission development strategies of the European Union

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(Resolution on the European Green Deal) [11] and its member states (including the Strategy of Latvia for the Achievement of Climate Neutrality by 2050 [12]). Therefore, international and national policymakers developing policy targets to limit GHG concentrations in the atmosphere, experts and institutions performing National GHG inventories, as well as policy implementers and forest managers require accurate data of past and current SOC stocks in forest soils and more knowledge to predict the potential future role of forest in GHG emissions and CO_2 sequestration [1,9,13]. A detailed review of the literature of the influence of forest management activities on SOC stocks and the key drivers and indicators for soil C stocks can be found in Mayer et al. (2020) and Wiesmeier et al. (2019), respectively [2,14].

The soil cover of the Baltic States is characterised by high diversity due to the varied composition of geological deposits and parent materials, diverse water conditions, and a comparatively large share of organic soils [15,16]. In Latvia, soils developed and evolved during the Holocene after the deglaciation of the territory are thus relatively young [16,17]. Forests are situated on soils formed on varying, mostly unconsolidated Quaternary deposits, and in some places on weakly consolidated pre-Quaternary terrigenous or hard carbonate sedimentary rocks [18]. Within the National GHG Inventory, the total reported forest area in Latvia (including afforested lands) was 3243.60 kha (50.2% of the total country area) in 2019 [19]. The distribution of organic soils in forest land is quantified based on the distribution of forest site types (data provided by the national forest inventory (NFI)) according to the national forest site type classification system [20], in addition to other ecosystem attributes, forest site typologies integrate soil types (organic or mineral) and soil moisture conditions (naturally dry, naturally wet or drained). Four forest site types with wet organic soils (upper organic soil or peat layers exceeding 30 cm thickness) and four forest site types with drained organic soils (upper organic soil or peat layers exceeding 20 cm thickness) are distinguished in Latvia [21], differing from one another in their distribution, structure, properties and the ways that they are used and managed. Forest site types with organic soil are linked to *Histosols* due to similar determination criteria [18], although the Intergovernmental Panel on Climate Change (IPCC) definition of organic soils [22] covers a much wider range of soils than the *Histosols* group [23].

In Latvia, drained organic soils in forest land (384.76 kha in 2019) are considered a key source of GHG emissions in the Land Use, Land Use Change, and Forestry (LULUCF) sector [19]. As Latvian forest typology is based on a combination of different ecosystem attributes and soil characteristics that may vary significantly within the boundaries of one compartment, the use of the distribution of forest site types to evaluate the area of organic soils in forest land in Latvia may introduce some error into the assessment of total GHG emissions from drained organic soils. The growing need to make recommendations for climate change mitigation measures in the LULUCF sector requires highly accurate evaluation of the SOC stock in forests and characterisation of the distribution of organic soils across different forest site types. This paper aims to describe the thickness of organic soil layers (O and H horizons) in all forest site types (both with mineral and organic soils) and to evaluate SOC stock in Latvian forests classified as land with organic soil to overall improve the National GHG Inventory.

2. Materials and Methods

2.1. Study Area

Our study was conducted in hemiboreal forests in Latvia. The hemiboreal zone is a transitional zone between the boreal and temperate forest of nemoral Europe, characterised by the coexistence of boreal coniferous species on poor soils and temperate broadleaved tree species on the most fertile soils [24]. According to data from the Latvian Environment, Geology and Meteorology Centre, the average annual air temperatures in the territory range from +5.2~+ 5.3 °C in the Alūksne and Vidzeme highlands to +6.8~+ 7.4 °C on the Baltic Sea coast. The warmest month of the year is July, with an average air temperature of +17.4 °C and an average maximum of +22.3 °C. February is the coldest month of the year, with an average air temperature of −3.7 °C and an average minimum air temperature of −6.6 °C. The annual precipitation in Latvia is 692 mm. The months with the highest precipitation are August and July, with averages of 77 and 76 mm, while the driest is April with an average of 34 mm.

2.2. Measurements of Soil Organic Layer Thickness in Forest Land

Soil organic layers were stratified into forest floor and peat layers (O and H horizons, respectively) according to the World Reference Base for Soil Resources (WRB) [25]. O horizon was defined as horizon dominated by organic material consisting of undecomposed or partially decomposed litter, such as leaves, needles, twigs, moss, and lichens, which has accumulated on the surface; it may be on top of either mineral or organic soils [25]. H horizon was defined as horizon dominated by organic material, formed from accumulations of undecomposed or partially decomposed organic material at the soil surface which may be under water; it may be on top of mineral soils or at any depth beneath the surface if it is buried [25].

The thicknesses of the O and H horizons were measured for 4599 NFI plots (Table 1) in forest land, evenly covering the whole country area in 2017–2019 (within the third cycle of the NFI). The thicknesses of the O and H horizons were measured at 4 points outside the plots: the measuring points were located approximately 1 m from the edge of the plot on the N, E, S, and W sides corresponding to azimuth angles of 0°, 90°, 180° and 270°. Measurements were made using a probe with a length of 70 cm. The thicknesses of the O and H horizons were measured using an undisturbed soil sample and ruler (accuracy 0.1 cm).

Table 1. Characteristics of plots in forest land where soil organic layer thickness was measured (NFI plots) and soil was sampled for physico-chemical analyses.

Soil Type and Moisture Conditions [1]	Forest Site Types [2]	Relative Soil Fertility [3]	Characteristics of NFI Plots [4] Where Thickness of Soil Organic Layers Was Measured			Soil Sampling [6]
			Number of NFI Plots	Average Age [5] (min–max)	Average Standing Volume ± S.E. (min–max), m^3 ha^{-1}	Number of Plots
Dry mineral soil	*Cladinoso–callunosa*	very low	42	70 (18–165)	163 ± 15 (10–466)	-
	Vacciniosa	low	148	67 (1–165)	213 ± 11 (<0.1–595)	-
	Myrtillosa	low	157	68 (1–170)	268 ± 14 (<0.1–696)	-
	Hylocomiosa	medium	818	53 (1–201)	259 ± 8 (<0.1–1123)	-
	Oxalidosa	above average	950	38 (1–182)	216 ± 6 (<0.1–1753)	-
	Aegopodiosa	high	151	56 (1–173)	264 ± 15 (<0.1–836)	-
Naturally wet mineral soil	*Cladinoso–sphagnosa*	very low	2	42 (31–53)	75 ± 50 (25–125)	-
	Vaccinioso–sphagnosa	low	73	53 (2–153)	140 ± 13 (<0.1–395)	-
	Myrtilloso–sphagnosa	medium	178	54 (1–193)	204 ± 13 (<0.1–780)	-
	Myrtilloso–polytrichosa	above average	154	45 (1–181)	187 ± 12 (<0.1–567)	-
	Dryopteriosa	high	11	47 (10–80)	248 ± 57 (7–525)	-
Drained mineral soil	*Callunosa mel.*	low	2	25 (24–25)	64 ± 20 (44–83)	-
	Vacciniosa mel.	medium	69	60 (1–141)	259 ± 20 (<0.1–645)	-
	Myrtillosa mel.	above average	511	48 (1–182)	247 ± 9 (<0.1–1046)	-
	Mercurialiosa mel.	high	236	40 (1–103)	223 ± 13 (<0.1–1458)	-
Naturally wet organic soil	*Sphagnosa*	low	137	76 (3–178)	88 ± 6 (<0.1–373)	13
	Caricoso–phragmitosa	medium	168	64 (1–168)	147 ± 8 (<0.1–445)	28
	Dryopterioso–caricosa	high	195	47 (4–143)	172 ± 10 (<0.1–643)	25
	Filipendulosa	high	8	57 (31–91)	243 ± 64 (28–523)	5
Drained organic soil	*Callunosa turf. mel.*	low	22	57 (27–210)	110 ± 14 (9–294)	13
	Vacciniosa turf. mel.	medium	102	67 (1–190)	202 ± 13 (<0.1–577)	17
	Myrtillosa turf.mel.	high	327	56 (1–195)	229 ± 10 (<0.1–759)	36
	Oxalidosa turf. mel.	high	138	44 (2–129)	208 ± 14 (<0.1–916)	37
Total	all	all	4599	51 (1–210)	220 ± 3 (<0.1–1753)	174

[1] Based on forest site type according to the national forest classification system [20]. [2] According to the national forest classification system [20]. [3] According to Kārkliņš et al. (2009) [17] based on the national forest classification system [20]. [4] Plots in forest land (excluding clear cut areas and afforested agricultural land). [5] Age of the dominant tree species in overstorey. [6] Soil sampling for physico-chemical analyses.

2.3. Soil Sampling and Analyses

For physico-chemical analyses, soil was sampled in 174 sample plots located in forest land with organic soil according to the national forest site type classification system [20] simultaneously meeting the organic soil criteria set by definition of IPCC [22]. O horizons were sampled separately using a square probe with an area of 100 cm^2. Fixed-depth sampling was applied to H horizon and mineral soil layers underlying the peat layer. Two replicates at 0–10 cm, 10–20 cm, 20–30 cm, 30–40 cm, 40–50 cm and 50–100 cm depth were taken using undisturbed soil sample probes (100 cm^3 volume steel cylinders) [26]. The 0 cm reference is at the top of the peat layer (H horizon) [26]. Soil sampling was carried out in 2012–2019.

Soil samples were prepared and analysed in the Laboratory of Forest Environment at the Latvian State Forest Research Institute 'Silava' following the reference methods outlined in Part X of the ICP Forests Manual on Sampling and Analysis of Soil [26]. The soil samples were prepared for analysis according to the LVS ISO 11464:2006 standard [27]. The following physico-chemical parameters were determined in the soil samples: soil bulk density (BD, kg m^{-3}) according to LVS ISO 11272:2017 [28], coarse fragments and fine earth fraction of soil (diameter (D) < 2 mm) according to LVS ISO 11277:2020 [29], total carbon (TC) concentration using elementary analysis (dry combustion) according to LVS ISO 10694:2006 [30], and carbonate concentration using an Eijkelkamp calcimeter according to ISO 10693:1995 [31]. The OC concentration (g kg^{-1}) in soil was calculated as the difference between TC concentration and inorganic carbon (carbonate) concentration. For chemical analyses, the fine earth fraction of soil (D < 2 mm) was used.

2.4. Soil Organic Carbon Stock Calculation

To compute the SOC stock in each individual organic soil layer down to 1 m depth (SOC_{LAY}, t C ha^{-1}), equation No. 1 was applied [3]:

$$SOC_{LAY} = (OC \times BD \times THICKNESS \times (1 - (P_{cf}/100)))/ucf, \quad (1)$$

where OC is the OC concentration in the fine earth of the layer, g kg^{-1}; BD is the soil bulk density, kg m^{-3}; THICKNESS is the layer thickness, cm; P_{cf} is the proportion of coarse fragments, %; and ucf is a unit correction factor of 10,000. The SOC stock below 1 m depth was not estimated.

To estimate SOC stock in forest land with organic soils at the national level, data on the distribution of forest site types in Latvia provided by NFI [32] were used.

2.5. Statistical Analysis

Data on soil organic layer thickness and SOC stock is pooled in groups according to forest site types, which integrate within themselves soil type (organic or mineral) and soil moisture conditions (naturally dry, naturally wet or drained) according to the national forest site type classification system [20]. Pairwise *t*-tests (pairwise comparisons using *t*-tests with pooled standard deviations (SD)) were used to evaluate differences in the thickness of soil organic layers and SOC stock between individual forest site types and pooled groups of forest site types according to soil types and moisture conditions. Correlations between the thickness of soil organic layers and stand characteristics were tested with Pearson's *r*. Both pairwise *t*-tests and Pearson's *r* were conducted using a significance level of $p < 0.05$. All statistical analyses were carried out using R [33].

3. Results

3.1. Thickness of Organic Soil Layers in Forest Land

NFI data shows that the thickness of the O horizon in forest land in Latvia ranged up to 20 cm (detected in *Myrtilloso–polytrichosa* stands dominated by black alder (*Alnus glutinosa* (L.) Gaertn.)). When differences between each individual forest site type (Figure 1) were compared, the highest average thickness of the O horizon was found in *Vaccinioso–sphagnosa* stands (3.4 ± 0.4 cm). When differences in the O horizon thickness between average values of groups of soil types and moisture conditions were compared, the highest average thickness of the O horizon (2.4 ± 0.2 cm) occurred in forests with wet mineral soil. Furthermore, the average thickness of the O horizon in forests with wet mineral soil was statistically significantly higher than in other groups of soil types and moisture conditions

($p < 0.001$). In forest land with mineral soil, the average thickness of the O horizon varies with soil fertility: the average thickness of the O horizon decreases with increasing soil fertility. Such a trend is not observed in forest land with organic soils.

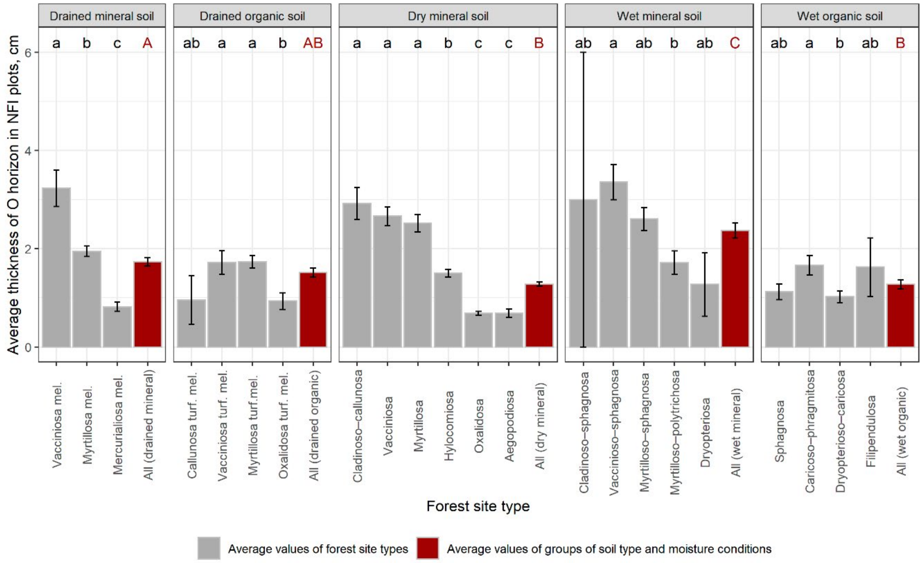

Figure 1. Average thickness of the O horizon in forest land in Latvia by forest site types. The division into groups based on soil types and moisture conditions is based on forest site types according to the national forest classification. Error bars represent standard errors. Different lower-case letters (black) show statistically significant differences ($p < 0.05$, $\alpha = 0.05$) in average values between forest site types within a group of soil type and moisture conditions; different upper-case letters (red) show statistically significant differences ($p < 0.05$, $\alpha = 0.05$) in average values between different groups of soil type and moisture conditions.

Figure 2 shows the average thickness of the O horizon in forest land in Latvia by dominant tree species. In forest land with mineral soil, the highest average thickness of the O horizon was detected in stands dominated by Scots pine (*Pinus sylvestris* L.) (2.7 ± 0.1 cm) followed by stands dominated by Norway spruce (*Picea abies* (L.) H.Karst.) (1.7 ± 0.1 cm). Furthermore, in forest land with mineral soil, the average thickness of the O horizon in stands dominated by Scots pine was statistically significantly higher than in stands with other dominant tree species ($p < 0.001$). In forest land with drained organic soil, the highest average thickness of the O horizon (2.1 ± 0.2 cm) was detected in stands dominated by Scots pine ($p < 0.022$) as well, but in forest land with wet organic soil, the highest average thickness of the O horizon (2.4 ± 0.6 cm) was detected in stands dominated by Norway spruce, furthermore, statistically significant difference between this and other dominant tree species ($p < 0.030$) was found.

No significant correlations were found between the thickness of the O horizon and forest stand characteristics such as basal area, standing volume, site index or age of the dominant tree species (Figure 3).

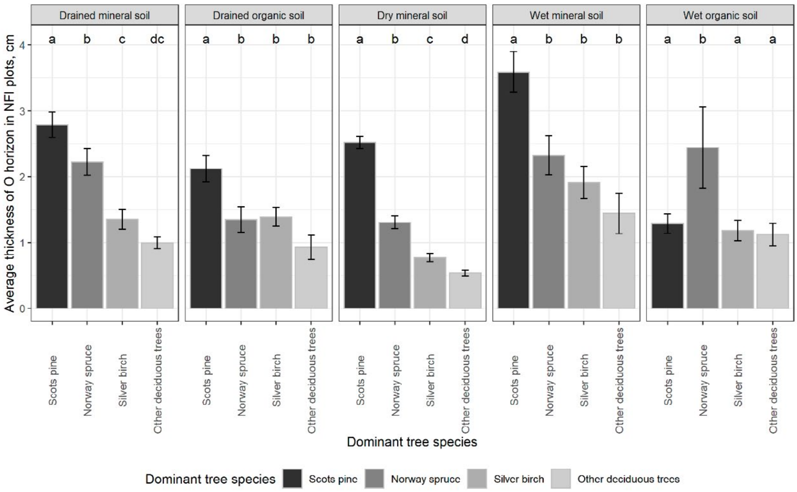

Figure 2. Average thickness of the O horizon in forest land in Latvia by dominant tree species (Scots pine (*Pinus sylvestris* L.), Norway spruce (*Picea abies* (L.) H.Karst.), silver Birch (*Betula pendula* Roth) and other deciduous trees). The division into groups based on soil types and moisture conditions is based on forest site types according to the national forest classification. Error bars represent standard errors. Different lower-case letters show statistically significant differences ($p < 0.05$, $\alpha = 0.05$) in average values between different dominant tree species within a group of soil type and moisture conditions.

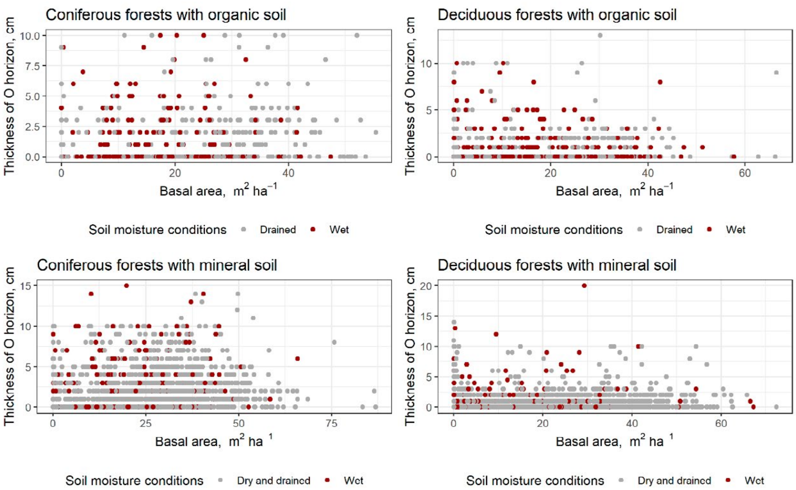

Figure 3. Distribution of thickness of the O horizon in forest land in Latvia depending on basal area. The division into groups based on soil types and moisture conditions is based on forest site types according to the national forest classification.

Figure 4 shows the proportion of the H horizon in the top 70 cm soil layer in forest land in Latvia by forest site type. As expected, a higher proportion of the H horizon in the top 70 cm was detected in land classified as forest land with organic soil ($p < 0.001$). If differences between individual forest site types are compared, the highest average proportion of the H horizon in the top 70 cm soil layer was detected in *Callunosa turf. mel.* stands characterised by drained organic soil (90 ± 6% of the top 70 cm soil layer). However, in general, a higher average proportion of the H horizon in the top 70 cm soil layer was detected in forests with wet organic soils (67 ± 2% of the top 70 cm) if compared with forests with drained organic soils (54 ± 2% of the top 70 cm). Furthermore, in forest land with organic soil (both in drained and wet conditions), the average proportion of the H horizon in the top 70 cm soil layer decreases with increasing soil fertility.

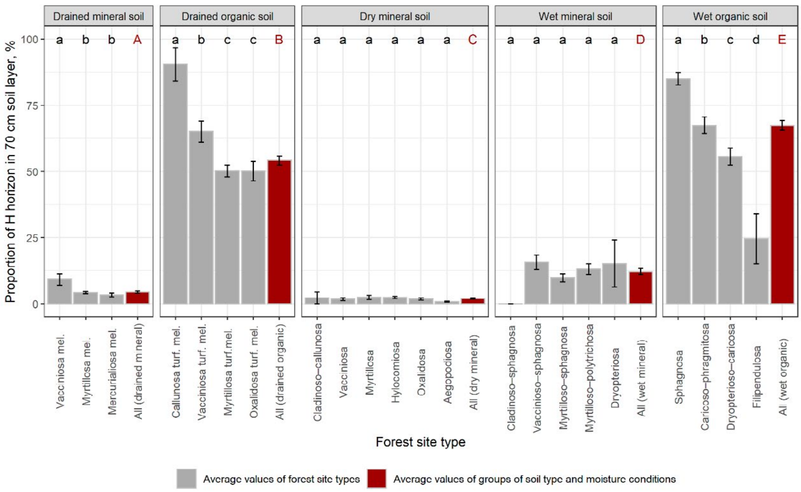

Figure 4. Proportion of the H horizon in the top 70 cm soil layer in forest land in Latvia. The division into groups based on soil types and moisture conditions is based on forest site types according to the national forest classification. Error bars represent standard errors. Different lower-case letters (black) show statistically significant differences ($p < 0.05$, $\alpha = 0.05$) in average values between forest site types within a group of soil type and moisture conditions; different upper-case letters (red) show statistically significant differences ($p < 0.05$, $\alpha = 0.05$) in average values between different groups of soil type and moisture conditions.

In forests with mineral soil, a statistically higher average proportion of the H horizon in the top 70 cm soil layer was detected in wet conditions (12 ± 1% of the top 70 cm) compared with forests with drained (4.4 ± 0.4% of the top 70 cm) and dry (2.0 ± 0.2% of the top 70 cm) mineral soils ($p < 0.001$).

In total, in forest land with mineral soil, the thickness of the H horizon was >20 cm in 3.5% of all NFI plots; relatively higher proportions were detected especially in wet mineral soils where the thickness of the H horizon was >20 cm, making up 12.9% of all NFI plots with wet mineral soils (Figure 5). In forest land with organic soils, as expected, the thickness of the H horizon was >20 cm in most NFI plots; nevertheless, in a relatively high proportion of NFI plots, the thickness of the H horizon was <20 cm (33.9% of all NFI plots with drained organic soils and 25.9% of all NFI plots with wet organic soils). The thickness of the H horizon was >70 cm in 0.6% of all NFI plots classified as plots with mineral soils, in 24.3% of all NFI plots classified as plots with drained organic soils and in 38.0% of all NFI plots classified as plots with wet organic soils (Figure 5).

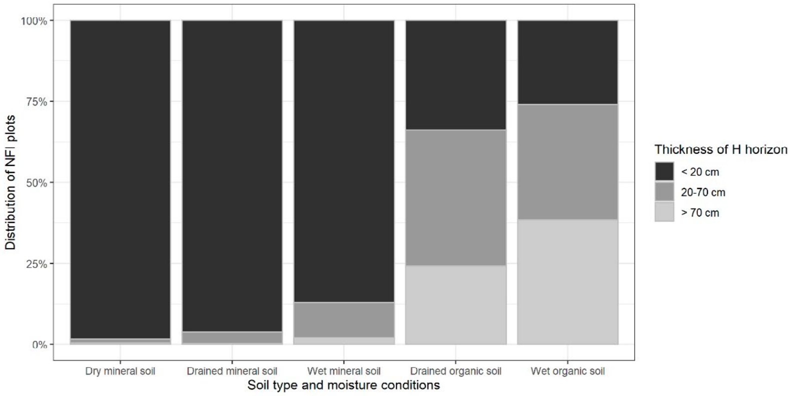

Figure 5. Distribution of NFI plots based on the thickness of the H horizon in forest land in Latvia. The division into groups based on soil types and moisture conditions is based on forest site types according to the national forest classification.

3.2. Soil Organic Carbon Stock in Forest Land with Organic Soil

In forest land with organic soil with an H horizon > 20 cm, the average OC concentration in the O horizon (Table S1) ranged between 490.6 g kg^{-1} (*Dryopterioso–caricosa*) and 554.9 g kg^{-1} (*Vacciniosa turf. mel.*). In the 0–20 cm soil layer, the average OC concentration variation was wider and ranged from 415.2 g kg^{-1} (*Oxalidosa turf. mel.*, 10–20 cm soil layer) to 539.7 g kg^{-1} (*Vacciniosa turf. mel.*, 10–20 cm soil layer). The average mass of the O horizon per area unit (Table S2) ranged from 12.7 g 100 cm^{-2} (*Filipendulosa*) to 45.0 g 100 cm^{-2} (*Sphagnosa*), but the average soil bulk density in the 0–20 cm soil layer ranged from 77.1 kg m^{-3} (*Sphagnosa*, 0–10 cm soil layer) to 302.5 kg m^{-3} (*Oxalidosa turf. mel.*, 10–20 cm soil layer).

Figure 6 shows the SOC stock per area unit in the O horizon, in the 0–30 cm layer, and in the 0–100 cm layer in forest land with organic soils (H horizon > 20 cm) in Latvia by forest site types. In the O horizon in forest land with wet organic soils, the forest site type average SOC stock ranged up to 23.9 $\pm$ 0.7 t C ha^{-1} in *Sphagnosa* stands (which had the lowest soil fertility in the group of wet organic soils). The weighted average SOC stock in the O horizon, which takes into account the distribution of forest site types in Latvia according to the NFI data, was 17.7 $\pm$ 2.3 t C ha^{-1}. In forest land with drained organic soil, forest site type average SOC stock varied up to 19.8 $\pm$ 2.8 t C ha^{-1} in *Myrtillosa turf. mel.* stands, while the weighted average SOC stock in the O horizon, considering the distribution of forest site types, was 17.4 $\pm$ 1.1 t C ha^{-1}.

In the 0–30 cm layer, the forest site type average SOC stock ranged up to 319.7 $\pm$ 21.9 t C ha^{-1} (in *Filipendulosa* stands), while the weighted average SOC stock in the 0–30 cm layer that considers the distribution of forest site types in Latvia according to the NFI data was 256.0 $\pm$ 7.8 t C ha^{-1} in drained organic soils and 189.3 $\pm$ 9.3 t C ha^{-1} in wet organic soils. Forest site type average SOC stock in the top 100 cm ranged up to 642.1 $\pm$ 91.3 t C ha^{-1} (also in *Filipendulosa* stands), and the weighted average SOC stock in the 0–100 cm layer was 546.5 $\pm$ 22.3 t C ha^{-1} in drained organic soils and 371.3 $\pm$ 20.9 t C ha^{-1} in wet organic soils.

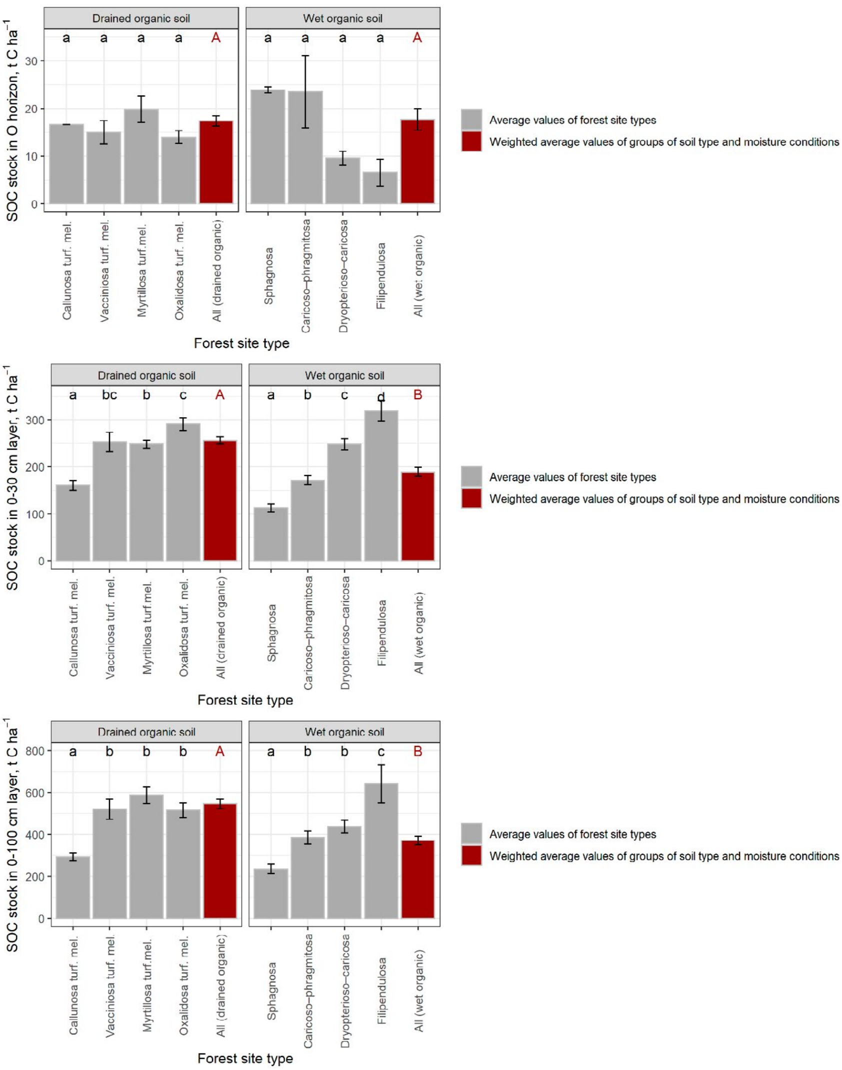

Figure 6. SOC stock per unit area in the O horizon, in the 0–30 cm layer and in the 0–100 cm layer in forest land with organic soils in Latvia. The division into groups of soil moisture conditions is based on forest site type according to the national forest classification. Error bars represent standard errors. Different lower-case letters (black) show statistically significant differences ($p < 0.05$, $\alpha = 0.05$) in average values between forest site types within a group of soil type and moisture conditions; different upper-case letters (red) show statistically significant differences ($p < 0.05$, $\alpha = 0.05$) in weighted average values between different groups of soil type and moisture conditions.

Both in the 0–30 cm soil layer and in the 0–100 cm soil layer, a statistically significantly higher average SOC carbon stock per area unit was found in drained organic soils if compared with forests with wet organic soils ($p < 0.001$). Furthermore, both in the 0–30 cm soil layer and in the 0–100 cm soil layer, average SOC stock increases significantly with increasing soil fertility, especially in forest land with wet organic soil (Figure 6).

Within the 0–100 cm layer, vertical SOC distribution showed that ~50% (ranging from 42 to 57%) of soil OC was stored in the upper 30 cm of the soil. The national-level assessment of SOC carbon stock in forest land with organic soils is summarised in Table 2.

Table 2. National-level assessment of soil organic carbon stock in the O horizon, in the 0–30 cm layer and in the 0–100 cm layer in forest land with organic soils in Latvia.

Soil Type and Moisture Conditions [1]	Forest Site Types [2]	Relative Soil Fertility [3]	Total Area in Latvia, Kha [4]	Soil Organic Carbon Stock, Mt C		
				O Horizon	0–30 cm	0–100 cm
Naturally wet organic soil	*Sphagnosa*	low	87.6	2.09	9.86	20.65
	Caricoso–phragmitosa	medium	105.5	2.48	18.06	40.72
	Dryopterioso–caricosa	high	137.3	1.32	34.07	60.17
	Filipendulosa	high	4.2	0.03	1.35	2.71
	total	-	334.6	5.92	63.34	124.25
Drained organic soil	*Callunosa turf. mel.*	low	17.2	0.29	2.77	5.05
	Vacciniosa turf. mel.	medium	68.4	1.03	17.32	35.58
	Myrtillosa turf.mel.	high	216.4	4.29	53.77	127.46
	Oxalidosa turf. mel.	high	99.0	1.38	28.83	51.12
	total	-	401.1	6.99	102.69	219.20
Total	all	-	735.7	12.90	166.02	343.45

[1] Based on forest site type according to the national forest classification system [20]. [2] According to the national forest classification system [20]. [3] According to Kārkliņš et al. (2009) [17] based on the national forest classification system [20]. [4] NFI data [32].

4. Discussion

Sequestration and storage of C in organic soil layers in forest land is currently discussed for many reasons, but recently the main emphasis has been on the achievement of climate change mitigation targets in the framework of international and national climate neutrality strategies by 2050.

4.1. Thickness of the O Horizon

The results presented in this study demonstrate that the O horizon thickness in coniferous forests is higher than in deciduous forests, with statistically significant differences were observed in all groups of soil type and moisture conditions except in wet organic soils. In Latvia, silver Birch (*Betula pendula* Roth; the dominant deciduous tree species in the country) has a slightly higher production rate of litter than coniferous tree species [34]. In the present study, production and decomposition of litter were not directly measured, but the thinner O horizon and lower mass of the O horizon per area unit in deciduous forests indicated faster decomposition of litter in the deciduous stands compared with the spruce and pine stands. Slower decomposition of coniferous litter can be explained by higher lignin content (e.g., [35,36]), although lignin concentrations vary within species (e.g., [37]). The soil moisture condition strongly influences the O horizon thickness. For instance, in forests with mineral soils, a statistically higher O horizon thickness was found in wet soils than in dry and drained soils both for coniferous and deciduous forests. This is related to a lower water table in dry and drained areas leading to an increase in the air-filled porosity of the organic matter layers, which in turn affects microbial processes and thus decomposition rates [38], whereas in wet soils decomposition is anaerobic and generally slow (e.g., [39]). In contrast, in forests with organic soil, a higher O horizon thickness was found in drained soils than in wet soils (although the difference was not statistically significant). This is explained by increased soil fertility after drainage [40] followed by increased tree biomass growth and higher litter production rates in drained soils [41,42] compensating for accelerated organic matter decomposition [38,43,44], as forest floor mass is the difference between litter accumulation (production) and decomposition [45]. The results presented in this study demonstrate that, in forest land with mineral soil, the average thickness of the O horizon decreases with increasing soil fertility. In addition, the differences in thickness and mass of the O horizon between stands in similar conditions can be explained by differences in the chemical composition of litter (soluble substances and labile compounds of litter are rapidly degraded, but cellulose and lignin decompose slowly [46]), root activity [7], bacteria and ectomycorrhizal fungal symbionts (e.g., [47]), microclimate, temperature (e.g., [48]) and presence of earthworms (e.g., [49]). When assessing the potential impact of climate change in the Baltic basin (higher annual average temperature and precipitation), it is hypothesised that changes in climate would result in higher N content in litter (a lower C/N ratio) and lower decomposition, and thus a considerable increase in organic matter accumulation [37].

A relatively large number of studies, both large-scale and regional, have found that the main drivers of forest litter production are climate (temperature and precipitation) and biomass abundance [50,51]. We tested correlations between the O horizon thickness and stand characteristics, but no significant relationships were found, although relationships between the litter production and stand characteristics such as basal area were previously found in hemiboreal regions [34]. This indirectly confirms the importance of decomposition rate on O horizon thickness and mass in forest land.

4.2. Thickness of the H Horizon

Although the national forest site type classification system states that forest land is classified as land with organic soils if the organic soil or peat layer is thicker than 30 cm in wet conditions and thicker than 20 cm in drained conditions [20], evaluation of the distribution of the H horizon thickness in NFI plots in forest land revealed that in 30% of forest land classified as land with drained organic soil, the H horizon was thinner than 20 cm, whereas in 4% of forest land classified as land with drained mineral soil, the H horizon was thicker than 20 cm. This is related to the unevenness of organic soil layer thickness in forest land; furthermore, previous forest soil research in Latvia has revealed that spatial distribution correlations do not always exist between forest site types, soil groups and prefix qualifiers according to the international WRB soil classification [18]. In addition, NFI plots are located in a regular grid regardless of major landforms, position and microtopography, and therefore soil at sampling points may not always be representative of the dominant soil type in the area as a whole.

As the specific IPCC definition of organic soils complies neither with the WRB soil classification nor with the Latvia Soil Classification, use of regular soil survey materials, to assess the area of organic soils within the National GHG Inventory, is either not possible or remains complicated [23]. In Latvia, within the National GHG Inventory, emissions from drained organic soils in forest land are estimated using NFI data on the area of drained organic soils based on the distribution of forest site types. In 2019, the total reported area of drained organic soils in forest land remaining forest land was 383.95 kha [19]. Taking into account the distribution of H horizon thicknesses estimated within this study, the corrected area of drained organic soil with an H horizon >20 cm was 274.67 kha in 2019 (less than reported in Latvia's National GHG Inventory by 28.5%). Within the Latvia's National GHG Inventory, GHG emissions from drained organic soils in forest land are estimated based on multiplying the area of organic soils by the relevant emission factors. Thus, overestimation of areas of organic soils in forest land could most likely reflect the overestimation of GHG emissions from drained organic soils in forest land in Latvia by approximately 360 kt CO_2 eq. (the sum of CO_2, CH_4 and N_2O emissions from soil and CH_4 emissions from drainage ditches calculated according to the methodology used in the Latvia's National GHG Inventory [19]) in 2019.

4.3. Soil Organic Carbon Stock in Forests with Organic Soils

In forest landscapes, variations in the determining factors of soil formation, i.e., parent material, topography, long-term interactions with organic matter input, organisms, dominant tree species, and climate and disturbances, result in large variability in SOC stocks [52]. De Vos et al. (2015) assessed SOC stocks based on data originating from 22 EU countries belonging to the UN/ECE ICP Forests Monitoring Level I network [3]. They estimated that the average SOC stock is 22.1 t C ha^{-1} in forest floors and 578 t C ha^{-1} in peat soils in the top 1 m [3]. In Latvia, Butlers and Lazdins (2020), in an earlier study of forests with organic soil, estimated that the largest values of OC stock in the O horizon were found in coniferous forests: up to 24.8 t C ha^{-1} in Norway spruce forests (~13 decades in age) and up to 20.5 t C ha^{-1} in Scots pine forests (~8 decades in age) [53]. They also concluded that C stock dynamics in the O horizon depend on the forest age according to polynomial regression, which demonstrates lower C stocks in young stands and an increase of C in mature forests with a subsequent decrease in decaying forests [53]. We calculated that the weighted average SOC stock in the O horizon, taking into account the distribution of forest site types in Latvia, was 17.7 $\pm$ 2.3 t C ha^{-1} in wet organic soils and 17.4 $\pm$ 1.1 t C ha^{-1} in forests with drained organic soil. The weighted average SOC stock in the 0–100 cm layer, considering the distribution of forest site types, was 546.5 $\pm$ 22.3 t C ha^{-1} in drained organic soils and 371.3 $\pm$ 20.9 t C ha^{-1} in wet organic soils. A higher soil bulk density and a lower proportion of the H horizon in the top 70 cm soil layer, but a higher SOC stock both in the 0–30 cm layer and in the 0–100 cm layer, were found in drained organic soils than in wet organic soils. This indicates a potential subsidence of organic matter caused mainly by physical shrinkage after drainage [43,54]. The weighted average soil bulk density in 0–10 cm soil layer in drained organic soils exceeded the soil bulk density in wet organic soils by 31 kg m^{-3}, and the difference between drained and wet organic soils increased up to 96 kg m^{-3} in 40–50 cm depth. These differences in soil bulk density resulted in higher weighted average SOC stock in the 0–100 cm soil layer in drained organic soils by ~175 t C ha^{-1} in total (97% of this value is due to differences in soil bulk density). It must be considered that SOC stock below 1-m depth was not estimated and this limits interpretations of the management (drainage) impact on SOC stocks in organic soils. In general, conclusions concerning drainage impact on SOC stock in organic soils in the boreal and hemiboreal vegetation zone are contradictory. For instance, Simola et al. (2009) reported a marked decrease of peat mass (C loses) in drained forestry peatlands in Finland [55]. Several other studies also carried out in the boreal and hemiboreal vegetation zone have revealed that SOC stock in forests with organic soils can remain stable or even continue to increase after drainage [42,43,56–58], but in warmer climate (temperate) regions, drained organic soil is mostly a net source of GHG emissions (e.g., [9]).

According to the results of this study, in Latvia, in forest land with organic soil (735.7 kha [32]), the total estimated SOC stock in the O horizon was 12.9 Mt C, but was 343.5 Mt C in the upper 100 cm soil layer. Butlers and Lazdins (2020) recently estimated that the total C stock in organic soil layers, including the litter layer and peat in the upper 70 cm soil layer (excluding potential C stock in mineral soil layers underlying the peat layer), in forests with organic soils in Latvia is 242 Mt C [53]. This indicates a considerable SOC stock stored under organic soil layers (litter and peat layers) up to 100 cm deep. The EU Forest Focus BioSoil study [59,60] approximated the total SOC stock in the O horizon and 0–80 cm soil layer in Latvia (at all forest site types both with mineral and organic soil) as ~754 Mt C [59]. Although SOC stock per unit area in forest land with mineral soil in Latvia is considerably lower (~ 195 t C ha^{-1} in the upper 80 cm [59]) than estimated within this study for forests with organic soil, most of the total SOC stock is located in forests with mineral soil, as forests with mineral soil cover most (2505.5 kha or 77%) of the total forest area in the country (3241.2 kha [32]).

5. Conclusions

In hemiboreal forests in Latvia, the highest average thickness of the O horizon was detected in coniferous forests with wet mineral soil. The average thickness of the O horizon in forests with mineral soil decreased with increasing soil fertility, but forest stand characteristics were weak predictors of O horizon thickness. By contrast, in forests with organic soil, higher O horizon thicknesses were found in drained soils than in wet soils, indicating that accelerated organic matter decomposition in drained soils [38,43,44] can be compensated by increased tree biomass growth followed by higher litter production rates [41,52] as a result of increased soil fertility after drainage [40].

In forests with drained organic soil, soil physico-chemical parameters (especially soil bulk density) indicate a potential subsidence of organic matter, caused mainly by physical shrinkage after drainage. Furthermore, distribution of the soil H horizon thickness across different forest site types highlighted the potential for overestimation of the area of organic soils and thus GHG emissions from drained organic soils in forest land in Latvia by approximately 360 kt CO_2 eq. in 2019 within the National GHG Inventory.

Supplementary Materials: The following are available online at https://www.mdpi.com/article/10.3390/f12070840/s1, Table S1: Organic carbon concentrations in soil in forest land with organic soil in Latvia, Table S2: Soil bulk density in forest land with organic soil in Latvia.

Author Contributions: Conceptualization, A.L.; methodology, A.L.; data curation, U.Z., R.P., A.J., A.A., and Ģ.R.; writing—original draft preparation, A.B. (Arta Bārdule); writing—review and editing, A.L., A.B. (Aldis Butlers) and I.L.; visualization, A.B. (Arta Bārdule); supervision, A.L.; project administration, I.L. All authors have read and agreed to the published version of the manuscript.

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Variation in Carbon Content among the Major Tree Species in Hemiboreal Forests in Latvia

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Abstract: This study was designed to estimate the variation in non-volatile carbon (C) content in different above- and belowground tree parts (stem, living branches, dead branches, stumps, coarse roots and small roots) and to develop country-specific weighted mean C content values for the major tree species in hemiboreal forests in Latvia: Norway spruce (*Picea abies* (L.) H. Karst.), Scots pine (*Pinus sylvestris* L.), birch spp. (*Betula* spp.) and European aspen (*Populus tremula* L.). In total, 372 sample trees from 124 forest stands were selected and destructively sampled. As the tree samples were pre-treated by oven-drying before elemental analysis, the results of this study represent the non-volatile C fraction. Our findings indicate a significant variation in C content among the tree parts and studied species with a range of 504.6 ± 3.4 g·kg^{-1} (European aspen, coarse roots) to 550.6 ± 2.4 g·kg^{-1} (Scots pine, dead branches). The weighted mean C content values for whole trees ranged from 509.0 ± 1.6 g·kg^{-1} for European aspen to 533.2 ± 1.6 g·kg^{-1} for Scots pine. Only in Norway spruce was the whole tree C content significantly influenced by tree age and size. Our analysis revealed that the use of the Intergovernmental Panel on Climate Change (IPCC) default C content values recommended for temperate and boreal ecological zones leads to a 5.1% underestimation of C stock in living tree biomass in Latvia's forests. Thus, the country-specific weighted mean C content values for major tree species we provide may improve the accuracy of National Greenhouse Gas Inventory estimates.

Keywords: living biomass; greenhouse gas inventory; Norway spruce; Scots pine; birch; European aspen

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1. Introduction

Forest ecosystems continuously exchange carbon dioxide (CO_2) with the atmosphere and are significant components of the global carbon (C) cycle [1–3]. In forests, living tree biomass is a key CO_2 sink due to the photosynthetic assimilation of CO_2 from the atmosphere [4,5]. During photosynthesis, atmospheric CO_2 is converted into carbohydrates and further integrated into the organic compounds that make up a plant's structure [4–6]. The durability and inertness of tree tissue maintain C in organic form over a relatively long period before it is returned to the atmosphere through respiration (oxidation of carbohydrates), decomposition or disturbance [4,5,7]. Worldwide since 2020, many countries have begun to count CO_2 sequestration and storage in living tree biomass in their national climate-change mitigation efforts as part of international climate policy agreements [2,8]. Thus, precise knowledge of the variation in C content of living tree biomass by species and biome is required to accurately quantify forest C stocks, validate forest C accounting models and support forest management strategies intended to maximize CO_2 sequestration [9–11].

In tree tissues, C is bound in organic compounds, mainly cellulose, hemicelluloses, lignin, extractable components and low molecular weight volatile compounds such as alcohols, phenols, terpenoids and aldehyde [9,10,12]. As the C content of these compounds varies considerably, the variation in total C content in tree tissues is largely determined by the proportions of these organic compounds. The proportions of organic compounds

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also vary widely among tree species and are affected by a particular tree's genetics and age, the location in the tree (including tissue type and proportion of heartwood to sapwood and earlywood or latewood), environmental and growth conditions such as climate, soil characteristics, sunlight and concentration of tropospheric ozone (O_3) and other factors [5,9,10,12–17]. The lignin content and ratio of lignin to cellulose are commonly considered the most important predictors of C content in tree tissues because lignin contains proportionally higher C content (up to 72% C) compared with other organic compounds [9,13,18].

An increasing number of studies conclude that the widely used assumption of 50% C content for all tree species and tissues as well as the simplified conversion factors recommended by the Intergovernmental Panel on Climate Change (IPCC) [19] may significantly over- or underestimate forest C stock in living biomass [5,9,10,13,15,18,20–22]. Thus, recommendations to use region-, species- and tissue-specific C fraction values aimed to reduce the uncertainty of forest C stock estimates are becoming increasingly important for calculations for National Greenhouse Gas (GHG) Inventories [9–11,20,23]. The importance of developing higher tier methods for calculating C turnover in the land use, land-use change and forestry (LULUCF) sector is determined by the targets set to transform the European Union (EU) economy and society to meet climate goals. According to the Revision of the Regulation on the inclusion of GHG emissions and removals in the LULUCF sector, Latvia must decrease GHG emissions in the LULUCF sector by more than 25 million tons of CO_2-eq (double the annual GHG emissions excluding LULUCF in Latvia in 2019) by 2030 and ensure continuous reduction of GHG emissions to compensate for emissions in the agricultural and other sectors before 2050. Meanwhile, the ageing of forests and disturbances caused by climate change is increasing pressure on forest ecosystems and tend to turn forests into a net source of GHG emissions. These processes require urgent action to ensure the resilient increase of forest C pools and to avoid increased GHG emissions from soils. Accurate and verified tools for modelling C turnover in forests are key to implementing the climate policy, particularly in the selection and projection of the effect of measures intended to reduce GHG emissions and increase CO_2 sequestration. Latvia's GHG inventory uses static (tree species and dimensions determined) biomass expansion factors and default IPCC values to estimate C content in biomass, which leads to potential over- or underestimation of C stock changes in living biomass and other C pools.

The main aims of this study were: (1) to evaluate variation in non-volatile C content across different above- and belowground parts of major tree species in Latvia (Norway spruce, Scots pine, birch and European aspen); (2) to develop country-specific weighted mean C content values for major tree species and species-dominated forest stands.

2. Materials and Methods

2.1. Study Area

Our study was conducted in the hemiboreal forests in Latvia. The hemiboreal zone is a transitional zone between the boreal and temperate forest of the nemoral zone in Europe characterised by the coexistence of boreal coniferous species on poor soils and temperate broadleaved tree species on fertile soils [24]. In total, 124 forest stands dominated by 4 different tree species (Norway spruce (*Picea abies* (L.) H.Karst.), Scots pine (*Pinus sylvestris* L.), birch spp. (mainly silver birch (*Betula pendula* Roth)) and European aspen (*Populus tremula* L.)) were selected. The selected forest stands represent different regions and tree populations in Latvia. In this study, we analysed material chosen to study the national biomass equations in Latvia [25]. In each of the selected forest stands, 3 sample trees representing the range of the dimensions of the dominant tree species in the stand were selected. Thus, the study material comprised a total of 372 sample trees (Table 1). Damaged and rotten trees were not accepted as sample trees. The collection of study material was performed from 2012 to 2014 during the dormant period when deciduous trees were leafless and young shoots had matured.

Table 1. Characteristics of sample trees.

Parameter, Unit	Value	Tree Species			
		Norway Spruce	**Scots Pine**	**Birch**	**European Aspen**
Number of sample trees	total number	81	102	105	84
Age [1], years	average range	41 9–97	54 6–141	35 8–92	23 5–76
Stem height, m	average ± S.E. range	16.6 ± 1.0 2.8–30.8	17.3 ± 0.9 1.9–34.5	18.1 ± 0.8 4.9–32.3	16.6 ± 0.9 3.7–29.9
Diameter at breast height, cm	average ± S.E. range	17.5 ± 1.0 2.4–36.3	19.0 ± 0.9 1.5–45.3	14.7 ± 0.7 2.7–37.2	13.8 ± 0.9 2.8–34.1

[1] Average tree age in stand.

2.2. Sampling Design and Chemical Analysis

A detailed sampling design for biomass estimation is described in Liepiņš et al. (2018) [25]. The biomass was estimated by individual tree part: stem, living branches (including needles for coniferous tree species), dead branches, stump and roots. Foliage as a separate tree part was not included in the analysis. Not all the biomass fractions were measured for all the sampled trees. In addition, there was a technical problem during sample pre-treatment, during which several samples were damaged and excluded from further analysis. The entire root system of the sampled trees was excavated manually for 145 trees. The total fresh weight of the stem and branches was measured in the field using field scales. The total weight of the stump and roots was determined in the spring or summer following tree felling.

After tree felling, the crown was divided into 3 sections of equal length, and one average-sized live branch was selected subjectively from the middle of each section. The 3 sample branches were weighed together in the field and sampled to determine the average moisture of the living crown. In addition, one average-sized dead sample branch per tree was collected from the lower part of the crown. After measuring the branches that were selected for subsequent dry weight determination, all remaining branches were cut off and weighed. The dry matter of the crown was calculated using the fresh to oven-dried weight ratio.

The stems were cross-cut into 1 or 2 m sections starting from the base of the stem and depending on the stem length (1 m sections for stems shorter than 20 m, 2 m sections for stems longer than 20 m). To calculate the dry stem biomass, sample discs were collected at the beginning of each stem section. Sample discs were also collected at the height of 1.3 m and the midpoint of the first section. The section biomass was calculated by multiplying the section mass by the section fresh to dry weight ratio calculated from the sample discs located at the ends of the stem sections; for the top section, however, only the base sample disc was used. The biomass of individual stem sections was summed to obtain the total stem biomass.

The entire root system of the sampled trees was excavated manually with hand tools to minimise the loss of the smallest roots. After root excavation and transportation to the processing location, the belowground parts were washed with a high-pressure water pump to remove all soil particles. To calculate the dry root biomass, each root system was divided into 3 sections:

- Stump—monolith (both above- and belowground portions), nondifferentiated parts of some roots;
- Coarse roots—diameter greater than or equal to 2 cm;
- Small roots—diameter less than 2 cm.

To calculate the dry weight of each belowground fraction, 1 sample disc was collected from the middle of the stump, 3 different diameter root discs were collected from the coarse roots, and 3 full-length roots less than 2 cm in diameter were collected to represent the small root biomass. The total belowground dry biomass was represented by the sum of the root fractions based on the individual fresh to oven-dry weight ratios of each part.

The dry weight of all samples was measured in the laboratory after drying at a temperature of 105 °C until a constant weight was reached.

For C content analysis, 1 medium-sized live branch (including needles for coniferous tree species), 1 dead branch and the belowground samples used for dry weight determination were used. In addition, 2 sample discs were collected from the stem at 1/6 and 2/3 of the total height (Figure 1). All samples used for dry weight and C content determination contained proportional shares of heartwood, sapwood and bark.

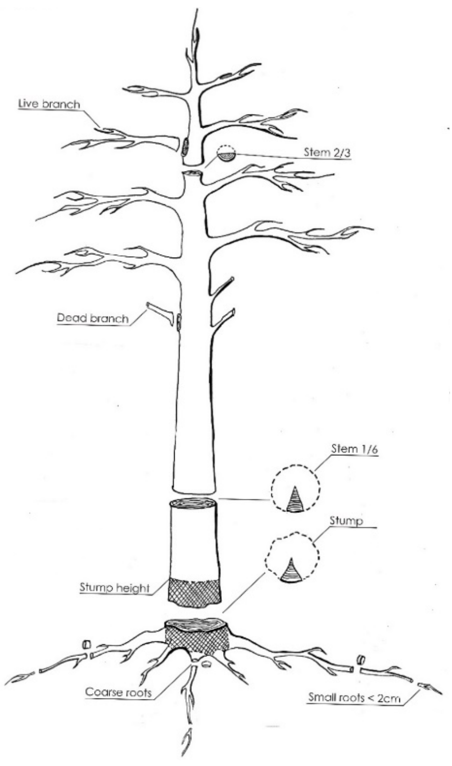

Figure 1. Tree part sampling design for C content analysis. All samples contained proportional shares of heartwood, sapwood and bark. Living branches of coniferous tree species contained proportional shares of needles, deciduous trees were leafless.

In the laboratory, each of the individual tree part samples (oven-dried at 105 °C until a constant mass was reached) was cut into small pieces and ground into a homogenous powder using Retsch SM 100 (Retsch GmbH, Hahn, Germany). Samples were analysed for non-volatile C content using the LECO CR-12 elemental analyser (LECO Corporation, St. Joseph, MI, USA) and recorded as C (g·kg^{-1}).

2.3. Data Analysis

The C content data were grouped according to tree species and part. Normality of data distribution was tested with Quantile-Quantile plots (QQ Plots) and Shapiro–Wilk tests, which approved that not all tested groups follow the normal distribution. Thus, non-parametric Kruskal-Wallis tests were used to evaluate differences in average C content values (including weighted mean C content values) between different tree parts or species. Correlation (Pearson's r) and regression analyses were used to quantify associations between the C content in different tree parts and several tree parameters (age, stem height, diameter at breast height). Both Kruskal-Wallis tests and Pearson's r were conducted with a significance level of $p < 0.05$. All statistical analyses were carried out with R [26].

Using the biomass and C content values of each tree part, the weighted mean C content (WMCC) for a single tree was calculated as follows [21]:

$$\mathrm{WMCC} = \frac{\sum(\mathrm{B_i} \times \mathrm{C_i})}{\sum \mathrm{B_i}} \times 100, \quad (1)$$

where B_i is the dry biomass of tree parts (kg tree^{-1}), C_i is the C content in tree parts (g·kg^{-1}) and i is the tree part.

The total C stock in living trees, including both above- and belowground tree parts in Latvian forests, was calculated using the species-specific weighted mean C content values determined within the present study (Table 2) and the National Forest Inventory (NFI) data (3rd cycle, 2014–2018) on tree biomass in forest land in Latvia. Values for tree species not included in the present study were estimated by type; the average weighted mean C content value of Scots pine and Norway spruce was used for other conifers, and the average weighted mean C content value of birch and European aspen was used for other deciduous tree species.

Table 2. Weighted means of C content in the tree for four main tree species in Latvia. Weighted means were calculated based on the proportional distribution of biomass of different tree parts. Different letters show statistically significant differences ($p < 0.05$) between different tree species within the same tree part.

Tree Part	Values	Weighted Mean C Content in Tree, g·kg^{-1}			
		Norway Spruce	Scots Pine	Birch	European Aspen
Aboveground parts	average ± S.E.	524.4 ± 1.4 [a]	530.4 ± 1.3 [b]	520.6 ± 1.4 [c]	510.2 ± 1.3 [d]
	median	524.2	531.3	520.4	509.8
	range	483.9–551.7	467.2–562.9	487.8–559.7	480.9–534.6
Belowground parts	average ± S.E.	529.9 ± 2.6 [a]	531.5 ± 2.4 [a]	527.9 ± 1.7 [a]	507.4 ± 2.1 [b]
	median	529.0	529.4	528.9	508.4
	range	497.2–559.3	486.5–567.0	502.9–549.6	482.1–531.9
Whole	average ± S.E.	526.5 ± 2.3 [a]	533.2 ± 1.6 [b]	521.4 ± 1.5 [c]	509.0 ± 1.6 [d]
	median	526.4	535.5	521.5	507.9
	range	489.8–546.2	502.1–554.7	501.3–550.5	490.0–527.4

Figure S1 shows the differences between C content values of different tree species estimated within the present study (WMCC) and the IPCC 2006 [19] or Martin et al. (2018) [18] values for temperate and boreal biomes.

3. Results

The C content in different tree parts varied significantly both within tree species (Figure 2) and across tree species (Figure 3). The mean C content in different tree parts of the studied tree species ranged from 504.6 ± 3.4 g·kg^{-1} (European aspen, coarse roots) to 550.6 ± 2.4 g·kg^{-1} (Scots pine, dead branches).

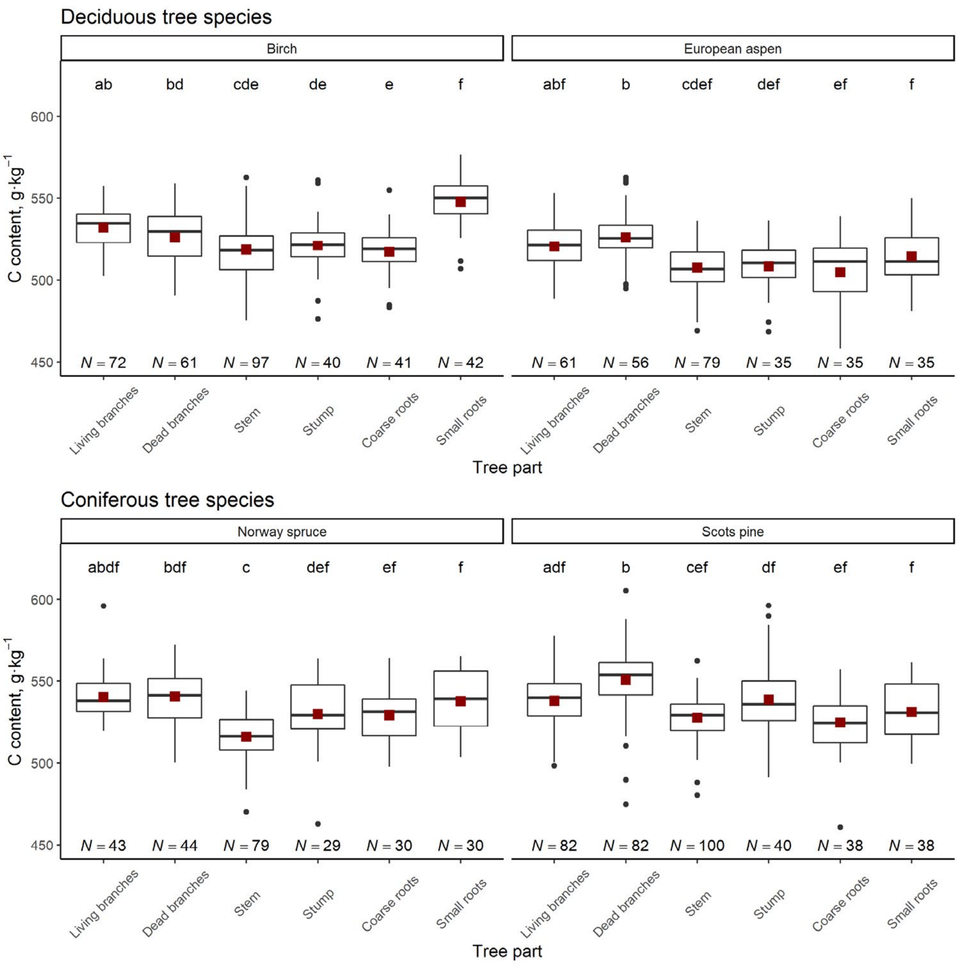

Figure 2. C content in different tree parts for the main tree species in Latvia. In the box plots, the median is shown by the bold line, the mean is shown by the dark red square, the box corresponds to the lower and upper quartiles, the whiskers show the minimal and maximal values (within 150% of the interquartile range from the median) and the black dots represent outliers of the datasets. Different letters show statistically significant differences ($p < 0.05$) between different tree parts within the same tree species. The number of samples (N) for each grouping is shown.

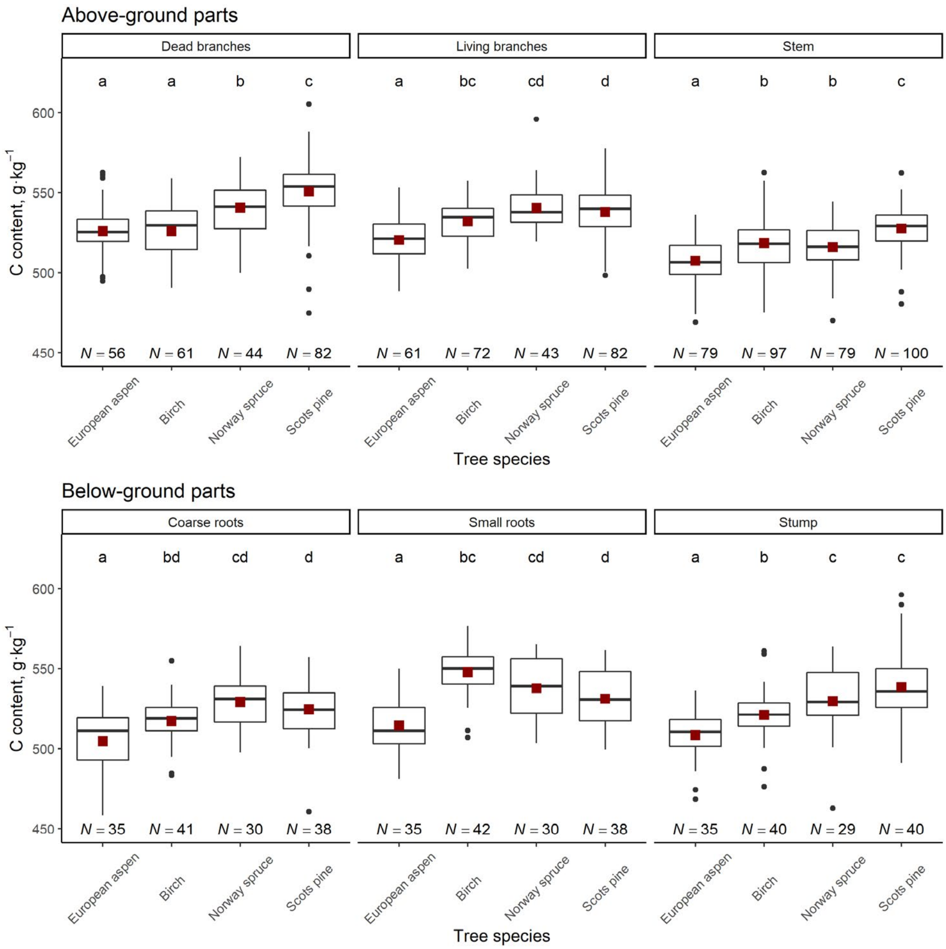

Figure 3. C content in different tree parts for the main tree species in Latvia. In the box plots, the median is shown by the bold line, the mean is shown by the dark red square, the box corresponds to the lower and upper quartiles, the whiskers show the minimal and maximal values (within 150% of the interquartile range from the median) and the black dots represent outliers of the datasets. Different letters show statistically significant differences ($p < 0.05$) between different tree species within the same tree part. The number of samples (N) for each grouping is shown.

In aboveground tree parts, living and dead branches were found to have the highest mean C content for all tree species, while in belowground tree parts, small roots were found to have the highest mean C content for all tree species except for Scots pine (Figure 2). The smallest difference between the C content of dead branches and stem was 0.8% in birch, whereas the largest difference was 2.5% in Norway spruce. The mean difference between the C content of living branches and stem varied in a slightly narrower range from 1.1% (Scots pine) to 2.5% (Norway spruce). Stumps and small roots tended to have higher C content than the stem for all tree species, reaching a maximum difference of 2.2% between the mean C content values of small roots and the stem in Norway spruce.

Among the studied tree species, the highest C content in dead branches, stem and stump was found for Scots pine; in living branches and coarse roots for Norway spruce and small roots for birch. European aspen showed the lowest mean C content in all tree parts (Figure 3).

Because the C content varied significantly between different tree parts (Figure 2), the weighted mean C content was calculated for each tree species (Table 2) based on biomass allocation in different tree parts. The largest weighted mean C content both in above- and belowground parts was found in Scots pine, while the lowest weighted mean C content was found in European aspen. More generally, conifers showed larger ($p < 0.001$) weighted mean C content compared with deciduous tree species: 527.7 $\pm$ 1.0 g$\cdot$kg^{-1} (N = 183) in conifers to 516.0 $\pm$ 1.0 g$\cdot$kg^{-1} (N = 189) in deciduous species for aboveground parts and 530.8 $\pm$ 1.8 g$\cdot$kg^{-1} (N = 67) in conifers to 518.9 $\pm$ 1.8 g$\cdot$kg^{-1} (N = 75) in deciduous specie for belowground parts.

Our estimated C content values were higher than the IPCC 2006 [19] or Martin et al. (2018) [18] values, and the greatest differences were observed when they were compared with Martin et al. (2018) [18] values for angiosperms in temperate biomes (Figure S1). The smallest difference (less than 2%) was observed for Norway spruce when compared with IPCC 2006 [19] values and for European aspen when compared with Martin et al. (2018) [18] values for boreal biomes (Figure S1).

A significant correlation ($r > 0.50$, $p < 0.05$) was found only between the C content of the stem at 1/6 and 2/3 of tree height as well as between the C content in the stump and coarse roots. In addition, correlation and regression analysis was used to identify the most influential variables affecting both tree part-specific and weighted mean C content. For Norway spruce, we found a moderate negative correlation between the C content of belowground parts (stump and coarse roots) and tree age and stem height (r values from -0.53 to -0.57, $p < 0.01$), but moderate positive correlations were found between the C content of small roots and tree age, stem height and diameter at breast height (r values of 0.54, 0.57 and 0.59, respectively, $p < 0.01$). For European aspen, a moderate negative correlation was found between the C content of dead branches and stem height ($r = -0.57$, $p < 0.001$), but for birch, moderate negative correlations were found between the C content of living branches and tree age, stem height and diameter at breast height (r values of -0.58, -0.62, -0.62, respectively, $p < 0.001$). In analyses of weighted mean C content values for each tree species separately, only the Norway spruce weighted mean C content of the whole tree was significantly influenced by tree age and size (Figure 4).

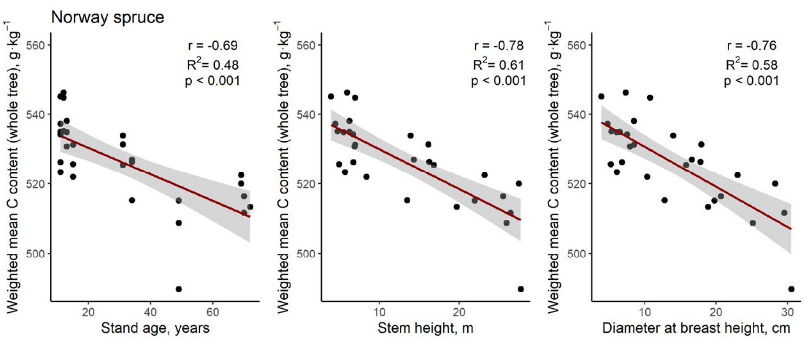

Figure 4. Relationships in Norway spruce between the weighted mean C content of the whole tree and tree age, stem height and diameter at breast height. Confidence interval is shown around smooth.

Considering the proportional distribution of the living biomass of admixed tree species in stands dominated by different tree species, weighted mean C content values for forest stands were developed (Table 3). This confirmed that the effect of admixed tree species on average weighted means of C content was negligible—the difference in weighted mean C content values between dominant tree species and forest stands dominated by those species was 0.01% for birch, 0.2% for Norway spruce, 0.4% for Scots pine and 0.7% for European aspen, respectively.

Table 3. Weighted means of C content for forest stands dominated by Norway spruce, Scots pine, birch or European aspen in Latvia. Weighted means were calculated based on the proportional distribution of whole tree biomass of different species in forest stands, taking into account the proportional distribution of admixed tree species. Different letters show statistically significant differences ($p < 0.05$) between stands with different dominant tree species.

Values	Weighted Mean C Content for Forest Stands, g·kg^{-1}			
	Stands Dominated by Norway Spruce	Stands Dominated by Scots Pine	Stands Dominated by Birch	Stands Dominated by European Aspen
Average ± S.E.	525.6 ± 0.1 [a]	531.3 ± 0.1 [b]	521.4 ± 0.1 [c]	512.7 ± 0.1 [d]
Median	526.4	532.1	521.4	512.0
Range	518.7–529.6	518.4–533.2	513.9–527.9	509.0–523.4

For all tree species other than coniferous tree species between 0 and 20 years old, most of the C in living trees was stored in stems followed by living branches. The percentage of C stock allocated to the stem trended higher with age for all tree species, reaching a maximum mean value of 77.5% of the total C stock in birches more than 60 years old. On the contrary, the C stock allocated to living branches trended lower with age for all tree species, with a maximum mean value of 53.2% found in Norway spruces 0 to 20 years old. The minimums mean value of 5.5% was found in birches more than 60 old. Similarly, the C stock allocated to small roots trended lower with age for all tree species; the highest mean value of 9.0% was found in birches 0 to 20 years old, and the lowest mean value of 1.9% was found in Scots pine more than 60 years old (Figure 5).

In forest land in Latvia covering 3472 thousand ha, including proportional shares of burned forest areas (0.06%), clear-cuts (1.39%), windrows (0.04%) and forested agricultural lands (10.39%), the estimated total C stock in living tree biomass was 251.6 Mt, including 198.5 Mt C in aboveground parts and 53.1 Mt C in belowground parts. The use of IPCC (2006) default C fraction values (48% for broad-leaved tree species and 51% for conifers [19]) may lead to an underestimation of the total C stock in living tree biomass in forest land in Latvia by 12.8 Mt C or 5.1%. The underestimation of C stock in living whole tree biomass using IPCC (2006) default C fraction values [19] may reach 18.8 t C ha^{-1} in forest stands dominated by Scots pine, 24.1 t C ha^{-1} in forest stands dominated by Norway spruce, 39.5 t C ha^{-1} in forest stands dominated by birch, and 27.1 t C ha^{-1} in forest stands dominated by European aspen.

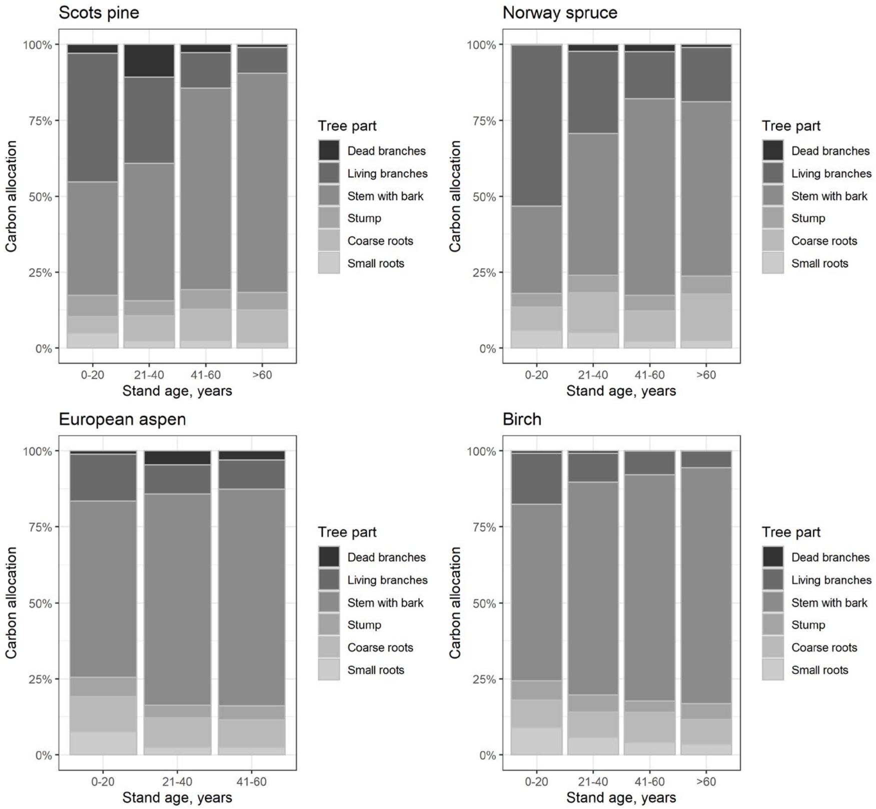

Figure 5. Carbon stock allocation in tree parts (including above- and belowground parts) of four main tree species.

4. Discussion

The total C content in tree tissues can be divided into fractions of non-volatile C and volatile C, in which the volatile C fraction consists of C compounds of low molecular weight [9,10]. As the tree samples were pre-treated by oven-drying before elemental analysis, the results of this study represent the non-volatile C fraction. We estimated that the weighted mean C content (whole tree) in the main tree species of hemiboreal forests in Latvia ranged from 50.9 ± 0.2% (European aspen) to 53.3 ± 0.2% (Scots pine), respectively. Although conifers showed statistically significantly higher weighted mean C content compared with deciduous tree species (53.0 ± 0.1% vs. 51.6 ± 0.1%), the variation in weighted mean C content within tree species exceeded the variation in weighted mean C content between species. Substantial variations in wood C content both among tree species as well as within individual trees were reported by both global level synthesis (e.g., [9,15,18]) and regional level studies (e.g., [5,21,27]). Our results were also consistent with previous findings that conifers have higher wood C content than deciduous tree species [5,9] and agree that the lignin content in the wood of conifer tree species is approximately 10% higher than that of deciduous tree species; lignin has the highest percentage of C compared with all other organic compounds in the wood [5,28].

Within recent global estimates of wood C content across the world's trees and forests, Martin et al. (2018) calculated that mean C content in trees (divided into conifers and angiosperm trees) in boreal and temperate forests ranged from 46.5% (angiosperm, temperate forests) to 50.1% (conifer, temperate forests) [18]. Ma et al. (2018), based on global estimates, reported that the mean C content in stem wood was 47.7% in deciduous broad-leaved trees and 50.5% in conifers but 46.6% and 48.4%, respectively, in roots [15]. Furthermore, they concluded that plant C content showed significant latitudinal trends induced by climatic factors and life forms [15]. Previously, Thomas and Martin (2012) estimated that, in temperate/boreal biomes, the wood C content across species ranged from 43.4% to 55.6%, but observed that the mean C fraction in stem wood was 48.8% in angiosperm tree species and 50.8% in conifer tree species [9]. The values Martin et al. (2018) [18], Ma et al. (2018) [15] and Thomas and Martin (2012) found for mean C content in the wood of boreal and temperate forests [9], as well as the IPCC 2006 values [19] (based on Lamlom and Savidge (2003) [5]), were lower than those estimated in this study. Nevertheless, species-specific and regional scale studies show C content values that are more similar to our estimates. For instance, Laiho and Laine (1997) in Finland reported C content values in different tree parts ranging from 51.8% (stem wood without bark) to 53.8% (foliage) for Scots pine; 50.9% (stem wood without bark) to 54.0 (foliage) for Norway spruce and 49.7% (stem wood without bark) to 55.7% (bark) for birch, respectively [29]. In north-western Turkey, the weighted mean C content in aboveground parts of Scots pine was found to be 52.0% [21], but, in Belgium, Janssens et al. (1999) reported C content in different tree parts of Scots pine ranging from 48.9% (stem) to 55.4% (fine roots) [27]. In North America, the mean C content of poplars (*Populus tremuloides* Michx. and *Populus trichocarpa* Torr. & Gray) wood was found to be 48.2% [5]. Gao et al. (2016) reported that the average total C content (sum of volatile and non-volatile C) in the stem wood and bark of the major tree species in the boreal forests of Canada were 50.5% and 56.2%, respectively [10]. When interpreting results and comparing C content values obtained in different studies for selected tree species both natural aspects (e.g., geographical location, climate, soil conditions, tree age, provenance, social position in the stand) and methodological nuances such as the sampling method (for instance, stem sample with or without bark), the selected sampling point on the tree, the sampling time (as the content of mobile C compounds varies by season), the sample pre-treatment method (for instance, oven-drying, ambient-temperature desiccating or freeze-drying) and the analysis method must be considered [5,9,10,13,21,30]. Thus, any comparison of C content values must be performed cautiously.

In comparing the C content of different tree parts, living and dead branches (wood with a proportional share of bark) were found to have the highest mean C content of aboveground tree parts for all tree species, while small roots were found to have the highest mean C content of belowground tree parts for all tree species except Scots pine, which showed the highest mean C content in the stump. Similarly, Tolunay (2009) [21] and Janssens et al. (1999) [27] reported that the highest C content in the aboveground parts of Scots pine was found in the branches, which aligns with our results. Furthermore, a trend of decreasing C content in branches by diameter was found [13]. In our study, stem bark and foliage (needles and leaves) were not included as separate tree parts, but several other studies showed relatively higher C content values in these parts in particular (e.g., [10,29,31]). This may be explained by a higher proportion of C-rich organic compounds, such as extractives, lignin and suberin, in stem bark compared with other tree parts [10,13]. For instance, Martin et al. (2015) [23] and Gao et al. (2016) [10] found extremely high bark C content in boreal paper birch (*Betula papyrifera*) (65.0 ± 3.6% and 60.7 ± 1.4%, respectively) and stressed that much of the variation in wood C content attributable to tissue type can be associated with variable C content in the bark. Furthermore, they revealed that the difference in C content between bark and stem wood was generally higher for boreal tree species than for temperate tree species [10,23]. Our results represent wood with a proportional share of bark for all tree parts.

The tree part-specific C content values obtained in the present study tend to show negative correlations with stand age and tree size (stem height and diameter at breast height). A similar pattern was found for Scots pine by Bert and Danjon (2006) [13], Bembenek et al. (2015) [30] and Wegiel and Polowy (2020) [31]. Tree age determines the sapwood to heartwood ratio as well as the proportional distribution of juvenile and mature wood [9,13]. Juvenile wood generally has a higher proportion of earlywood [30,32] and thus a higher extractive and lignin content than mature wood to support mechanical stability and defence mechanisms [10]. Juvenile trees, therefore, have a higher C content [5,9,13]. More recent findings by Gao et al. (2016) [10] and Martin et al. (2013) [33] highlighted that the tree age- and size-associated trend of total C content was likely led by variations in the proportion of volatile C compounds. Furthermore, they speculate that the amount of volatile C was the most important predictor of the overall variation in the total C content in trees [10,33]. Most importantly, increasing evidence shows that disregarding the differences in C content among different tree parts as well as the size- and age-dependent changes in C content in tree biomass could lead to errors in estimating the C stock in living tree biomass (e.g., [31,34]).

Along with other factors, tree age strongly determines the total C stock in living tree biomass and the allocation of C stock across different tree parts [27]. The results of this study showed that the relative contribution of living branches and small roots decreased with tree age for all tree species, but the contribution of the stem trended higher with tree age, reaching 77.5% of the total C stock in birches more than 60 years old. The C stock sum of all aboveground parts ranged from 74.4% in European aspens 0 to 20 years old to 84.4% in Scots pines 21 to 40 years old, and the highest C stock of belowground parts was found in young European aspens (25.6%), with the lowest found in Scots pines 21 to 40 years old (15.6%). In general, our estimates of C stock distribution across tree parts fell within the ranges reported by previous studies (e.g., [21,27,29,35]).

Other studies have reported a much wider range of tree species- and tree part-specific C content values for different biomes [9,15,33] than the default IPCC (2006) values [19]. Thus, the use of the default IPCC (2006) C content values may over- or underestimate C stock in living tree biomass. Our study shows that using the default IPCC (2006) C content values to estimate C stock in the living biomass of forest land in Latvia may lead to an underestimation of 5.1% or 12.8 Mt C.

In forest stands dominated by Norway spruce, Scots pine, birch or European aspen, the admixture of other tree species is common in hemiboreal forests. A combination of weighted mean C content values for each tree species and NFI data (3rd cycle) showed that the average proportion of C stock in living biomass formed by admixed tree species ranged from 17% in forest stands dominated by Norway spruce to 30% in forest stands dominated by European aspen. Customised weighted mean C content values were developed for forest stands dominated by Norway spruce, Scots pine, birch or European aspen considering the admixture of other tree species (Table 3). The difference in weighted mean C content values for living tree biomass between dominant tree species and forest stands dominated by those species reached 0.7% for European aspen (509.0 $\pm$ 1.6 vs. 512.7 $\pm$ 0.1 g$\cdot$kg^{-1}). The difference for stands dominated by birch, Norway spruce and Scots pine was even more negligible (<0.4%).

5. Conclusions

The results of this study provided tree part-specific and weighted means of C content values for the main tree species in Latvia. Statistically significant C content variation was found among different tree parts as well as among tree species with a range of 504.6 $\pm$ 3.4 g$\cdot$kg^{-1} (European aspen, coarse roots) to 550.6 $\pm$ 2.4 g$\cdot$kg^{-1} (Scots pine, dead branches). Weighted mean C content values based on proportional biomass distribution of different tree parts for each tree species are recommended to increase the accuracy of C stock in living tree biomass estimates in the National GHG Inventory (weighted mean C content values for whole trees: 526.5 $\pm$ 2.3 g$\cdot$kg^{-1} for Norway spruce, 533.2 $\pm$ 1.6 g$\cdot$kg^{-1} for Scots pine, 521.4 $\pm$ 1.5 g$\cdot$kg^{-1} for birch and 509.0 g$\cdot$kg^{-1} for European aspen). Furthermore, the results highlight that using the default IPCC C content values [19] results in underestimation of the C stock in living tree biomass in Latvia.

Supplementary Materials: The following are available online at https://www.mdpi.com/article/10.3390/f12091292/s1, Figure S1: the difference in C content of different tree species between values as estimated in this study (weighted means) and IPCC 2006 values for temperate/boreal biomes or Martin et al. (2018) values for temperate and boreal biomes separately.

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Estimation of litter input in hemiboreal forests with drained organic soils for improvement of GHG inventories

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Abstract

Assessments of net greenhouse gas (GHG) emissions in forest land with drained organic soils conducted within the scope of National GHG Inventories require reliable data on litter production and information on carbon (C) input to soil. To estimate C input through tree above-ground litter, sampling of above-ground litter was done in 36 research sites in Latvia representing typical forests with drained organic soils in the hemiboreal zone. To estimate C input through tree below-ground litter and litter from ground vegetation, modelling approach based on literature review and data on characteristics of forest stands with drained organic soils in Latvia provided by the National Forest Inventory (NFI) was used. The study highlighted dependence of C input to soil through litter production on the stand characteristics and thus significant differences in the C input with litter between young and middle-aged stands. The study also proved that drained organic soils in the middle-aged forests dominated by silver birch, Scots pine and Norway spruce may not be the source of net GHG emissions due to offset by C input through litter production. However, there is still high uncertainty of C input with tree below-ground litter and ground vegetation, particularly, mosses, herbs and grasses which may have crucial role in C balance in forests with drained organic soils.

Keywords: forests, drained organic soils, litter production, carbon input, National GHG Inventory

Introduction

Worldwide, organic soils have large carbon (C) and nitrogen (N) stores, and they can both remove and emit greenhouse gases (GHGs), thus contributing to the atmospheric GHG concentrations (Jauhiainen et al. 2019, Ziche et al. 2019). Organic soils are formed from partially decayed plant remains in anaerobic conditions through generally slow accumulation and compaction below the high water-table (WT) in peat-forming ecosystems (Moore 1989, Jauhiainen et al. 2019). Organic soil layer accumulation depends on the equilibrium between production and decay of organic matter that is highly sensitive to major climate change and management impacts (Joosten 2015). In the Nordic and Baltic countries, peat-forming ecosystems have been widely converted into forest land (Paavilainen and Päivinen 1995, Jauhiainen et al. 2019). These land use changes commonly involve drainage by ditching to promote forest growth, but it changes soil conditions enhancing mineralization of organic matter under aerobic conditions and results in activation of soil C and N stores (Jauhiainen et al. 2019). Drainage diminishes the emission of methane (CH_4), but simultaneously increases emissions of carbon dioxide (CO_2) and nitrous oxide (N_2O) from soil. In addition, drainage ditches itself are a large source of CH_4 emissions and carry dissolved organic carbon (DOC) and other C-forms out of the ecosystem, which is then largely emitted off-site as CO_2. Furthermore, deeper drainage and warmer climates increase emissions from organic soils (Joosten 2015). Globally, 15% of the organic soils are drained (Joosten 2015), but in Europe even 48% of the organic soils are drained, especially in the temperate zone (RRR 2017). Although drained organic soils comprise about 0.4% of the global land area, these soils contribute significantly (~5%) to global anthropogenic GHG emissions (Joosten 2015).

Within the National GHG Inventory reports under the United Nations Framework Convention on Climate Change (UNFCCC) and the Kyoto Protocol, anthropogenic CO_2, CH_4 and N_2O emissions from organic soils in forest land are reported under the Land use, Land use change and Forestry (LULUCF) sector (IPCC 2006, Tiemeyer et al. 2020). Although organic soils have a large impact on the total GHG budget in the LULUCF sector (Lazdiņš and

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Lupiķis 2019) and there are growing international requirements for improved accuracy of estimates of CO_2 removals and GHG emissions from organic soils (IPCC 2014), annual GHG emission factors (EFs) from organic soils are still characterized by high uncertainty rate (Jauhiainen et al. 2019) and significant differences between regions even in the same climate zone (Lazdiņš and Lupiķis 2019).

After drainage of organic soils EFs reflect the impact of climatic conditions on the decomposition rate of organic matter. Consequently, moving from higher to lower latitude, emissions from drained organic soils increase (Biancalani and Avagyan 2014). This explains the differences between the IPCC default EFs for temperate climate/vegetation zone calculated on the basis of results obtained in the central and northern parts of Europe and recent findings in Latvia located in the hemiboreal zone – the transitional zone between the boreal and temperate forests of nemoral Europe. For instance, Lupiķis and Lazdiņš (2017) estimated that emissions from drained organic soils in forest land in Latvia equal 0.52 t CO_2-C ha^{-1} yr^{-1}, but the IPCC default EF for temperate climate/vegetation zone is significantly higher – 2.6 t CO_2-C ha^{-1} yr^{-1}. Similarly, research in deciduous and coniferous forest stands in extracted peat fields in Latvia (LIFE REstore 2020) reflected that the IPCC default EF given in the 2013 IPCC guidelines most probably overestimate emissions from organic soils in Latvia (Lazdiņš and Lupiķis 2019).

Litter production is a key parameter in estimating, modelling and predicting forest soil organic carbon (SOC) stocks and its changes responding, for instance, to management practices or climate change (Wutzler and Mund 2007, Hansen et al. 2009, Cao et al. 2019, Feng et al. 2019). Thus, GHG assessments would benefit from reliable litter production information (Neumann et al. 2018). Soil organic matter is primarily plant-derived, contributing to the accumulation of SOC due to humification after plant death, or root-borne organic substances released into the rhizosphere during the plant growth (Kuzyakov and Domanski 2000). It is important to quantify contributions from both above-ground inputs and below-ground inputs to understand the amount of C ultimately stored in the soil (Ekberg et al. 2007, Cao et al. 2020).

Although it is considered that C input through above-ground litter is well investigated (Kuzyakov and Domanski 2000), reports on relationship between inputs of plant above-ground litter and SOC dynamics are still in controversy. Numerous studies have been done to estimate regional drivers of litter production using both field measurements of litter production and modelling approaches (e.g. Wutzler and Mund 2007, Hansen et al. 2009, Becker et al. 2018, Cao et al. 2019, Ziche et al. 2019). Although forest ecosystems are highly complex and various factors exert large spatial heterogeneity (Qin et al. 2019), there are some large-scale efforts to develop litter production models and determine total litter contribution to C cycling in forests addressing climat-, region- and species-specific differences, and its temporal trends. For instance, Liu et al. (2004) determined the relationships between climatic factors and litter production in forests of Eurasia. The results indicate that annual mean temperature has a greater effect on litter production compared to the annual precipitation across Eurasian forests. Furthermore, the results highlighted a difference in climate control between coniferous and broadleaf forests at a continental scale, and consequently different litter production responses to climate change (Liu et al. 2004). Based on data obtained within pan-European forest monitoring of the International Co-operative Programme on Assessment and Monitoring of Air Pollution Effects on Forests (ICP-Forests), Neumann et al. (2018) recently improved existing litter production estimation models that require climate information (Liu et al. 2004) by adding biomass abundance approach (leaf area index and stand density index) to quantify litter fluxes aggregated by bioregions and by forest types across Europe. In Latvia, continuous data on litter production in forests is available from the ICP-Forests Level II monitoring plots located in Scots pine stands, but this data set represents forest stands only on dry mineral soils.

While it is relatively easy to collect above-ground litter and estimate its production, quantification of below-ground litter still remains a challenge. The below-ground litter consists of dead roots, mycorrhizae and root exudates. Fine roots are commonly defined as non-woody, short-lived roots that are 2 mm or less in diameter and they represent one of the largest fractions of below-ground litter (Lehtonen 2005, Clemmensen et al. 2013, Leppälammi-Kujansuu et al. 2014, McCormack et al. 2015). The fine root turnover rate is a number that represents the times fine root biomass is replaced annually (Hendrick and Pregitzer, 1992). The fast turnover rates of fine roots ensure a major long-term contribution to below-ground C stocks, although fine root biomass constitutes less than 5% of the total forest biomass (Vogt et al. 1996). The direct methods to measure fine root turnover are ingrowth cores and minirhizotrons, whereas the indirect methods are C isotopic measurements, sequential soil coring, N budget, C budget and correlations with abiotic resources (Lukac 2012, Yuan and Chen 2012).

Excavation of roots for direct measurements is labour-intensive, changes the natural environment and causes artefacts, so that the measurements are no longer fully representative. Therefore, modelling approaches are widely used instead of field measurements to determine fine root biomass and turnover from other easily measurable stand variables. The input data most often include foliage and above-ground biomass, leaf area index (LAI), climate, latitude, net primary production, and land cover type (Liski et al. 2002, Liu et al. 2004, Härkönen et al. 2011, Yuan et al. 2018). The main principle of allometry is that for trees growing under the same conditions there are certain proportions between their dimensions, e.g., height and diameter, biomass and diameter. This principle can be used to predict one variable from another, using allometric equations. Remotely sensed information from satellites or inventory-based gridded forest data also can be applied to predict fine root characteristics at large scale (Yan et al. 2016, Moreno et al. 2017). Above-ground litter values can also be used to estimate below-ground litter. Chen et al. (2018) extended the pipe model analysis proposed by Shinozaki et al. (1964) and estimated that the ratio of fine root production against leaf production at the stand level is about 0.8. Fine root biomass also correlates positively with stand basal area (Vanninen and Mäkelä 1999, Helmisaari et al. 2007, Finér et al. 2011, Lehtonen et al. 2016).

Ground vegetation is another important yet less studied component of forest ecosystems. The C budgets of trees and forest soil have been modelled extensively, but vegetation is usually excluded from these analyses. According to studies carried out in pine and spruce upland forest stands in Finland, ground vegetation comprises about 4–13% of the C stock (Mälkönen 1974, Havas and Kubin 1983). Other studies show that the proportion of the C stock in ground vegetation is 1–2% (Lakida et al. 1996, Pussinen et al. 1997). Although ground vegetation constitutes only a small proportion of biomass in forests, it contributes significantly to nutrient cycles because of the fast turnover and easily decomposable litter (Mälkönen 1974, Palviainen et al. 2005). Consideration of ground vegetation biomass is particularly important during the early-successional stages of forest after clear-cutting or fire disturbances, when it is the main living vegetation component (Palviainen et al. 2005). Ignoring ground vegetation may lead to underestimation of net primary productivity, litter production and the C stock of soil. Biomass of ground vegetation decays and regenerates rapidly, therefore removals in biomass re-growth balance the emissions from decay. In peatlands the proportion of ground vegetation is mainly influenced by the WT level and the structure of the tree layer (Finér and Nieminen 1997, Minkkinen et al. 1999).

There are several methods to estimate ground vegetation biomass. The point-intercept method determines the number of contacts between plants by passing a pin through the vegetation at many positions (Levy and Madden 1933, Goodall 1952). This method gives highly accurate biomass estimates; however, it is destructive, labour-intensive and not suitable for large-scale inventories. Percentage cover analysis is a non-destructive alternative that can be applied extensively; however, it is less accurate, due to differences in visual estimates of each observer. Several studies show a relation between the percentage cover and biomass (Chiarucci et al. 1999, Röttgermann et al. 2000). Muukkonen and Mäkipää (2006) developed equations for pine, spruce and broad-leaved forest stands to calculate ground vegetation biomass using stand age and site attributes. There are models for specific vegetation types such as dwarf shrubs, herbs and grasses, mosses, lichens, total field layer, total bottom layer and all ground vegetation together. Models, where only stand age is an explanatory variable, can also be used in other boreal countries. Stand age is considered a significant predictor of ground vegetation because of the influence of structural changes stands undergo during their development. Light availability changes along with leaf area index, and there are shifts in vegetation from heliophilous species (herbs and grasses) towards species adapted to shady environments (e.g. mosses) as well as changes in abundance and occurrence of certain species (Lindholm and Vasander 1987, Luyssaert et al. 2007).

The specific aim of the study was to contribute to improvement of knowledge on C input to soil through plant litter production, including tree above- and below-ground litter and ground vegetation litter in the hemiboreal region (Latvia is a target area) to generally improve the National GHG Inventory.

Materials and methods

Tree above-ground litter collection and analysis

We conducted the study in central Latvia. Sampling of tree above-ground litter was performed in 36 research sites representing typical forests with drained organic soils in the hemiboreal region (Figure 1). The forest site types based on Bušs (1981) in the order from relatively nutrient poor to nutrient rich soils (Kārkliņš et al. 2009) are: *Callunosa turf. mel.* (relatively low soil fertility), *Vacciniosa turf. mel.* (moderate soil fertility), *Myrtillosa turf. mel.* (relatively high soil fertility), and *Oxalidosa turf. mel*. (relatively very high soil fertility). The research sites were dominated by Scots pine (*Pinus sylvestris* L.), Norway spruce (*Picea abies* (L.) H. Karst.), or silver birch (*Betula pendula* Roth). The mean annual precipitation in the study region was 732 mm and the mean annual temperature was 8.1 °C in 2019 (calculated as average using data obtained from two nearest observation stations in Sigulda and Skriveri; Latvian Environment, Geology and Meteorology Centre). More detailed characteristics of the research sites are presented in Table 1.

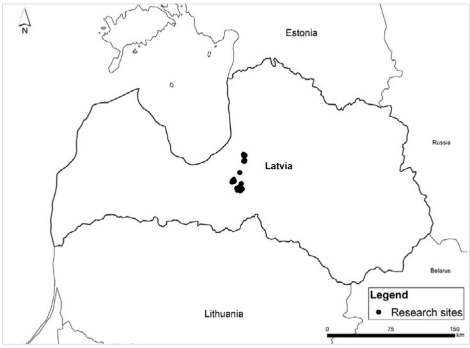

Figure 1. Location of the research sites (forest stands with drained organic soils) in Latvia

Table 1. Characteristics of the research sites located in the typical forests with drained organic soils in the hemiboreal zone in Latvia

Forest site type	Soil type	Dominant tree species (number of research sites)	Number of trees per hectare, count ha^{-1}	Diameter, cm	Height, m	Basal area, $m^2\ ha^{-1}$	Stock, $m^3\ ha^{-1}$	Age, years
Vacciniosa turf. mel.	Fibric histosols	Scots pine (2)	550 ± 30	22.2 ± 0.8	21.5 ± 0.1	22.4 ± 0.3	238 ± 4	90 ± 5
			(520–580)	(21.4–23.0)	(21.3–21.6)	(22.1–22.8)	(234–241)	(85–95)
Oxalidosa turf. mel.	Terric histosols	silver birch (8)	1,488 ± 295	15.2 ± 1.9	16.5 ± 1.3	26.5 ± 3.5	264 ± 50	47 ± 8
			(360–3,080)	(8.6–21.8)	(10.2–20.6)	(16.8–42.7)	(138–500)	(22–68)
		Norway spruce (10)	1,000 ± 141	20.0 ± 1.4	18.7 ± 1.0	32.9 ± 2.4	348 ± 40	51 ± 5
			(620–2,040)	(10.3–27.7)	(12.1–23.6)	(20.7–44.5)	(153–586)	(26–78)
Myrtillosa turf. mel.	Terric histosols	silver birch (5)	1,400 ± 135	14.1 ± 1.5	15.9 ± 1.1	24.6 ± 4.0	229 ± 49	60 ± 5
			(1,120–1,860)	(10.8–19.4)	(13.1–18.6)	(15.9–40.0)	(115–407)	(45–70)
		Norway spruce (1)	1,100	10.2	9.8	10.5	68	59
		Scots pine (3)	693 ± 216	23.0 ± 4.6	19.1 ± 4.5	28.2 ± 6.3	310 ± 122	63 ± 22
			(420–1120)	(14.2–29.7)	(11.3–26.9)	(18.6–40.1)	(112–533)	(23–98)
Callunosa turf. mel.	Fibric histosols	silver birch (3)	2,047 ± 704	9.5 ± 2.4	11.4 ± 1.6	13.1 ± 1.2	90 ± 17	30 ± 0
			(840–3,280)	(6.3–14.1)	(9.5–14.7)	(11.5–15.5)	(67–125)	(30–31)
		Scots pine (4)	1,460 ± 214	10.6 ± 1.8	9.7 ± 1.7	14.5 ± 3.9	93 ± 32	42 ± 12
			(980–2,020)	(6.6–13.8)	(5.4–12.6)	(5.3–23.8)	(21–163)	(21–70)
Average	-	silver birch (16)	1,565 ± 195	13.8 ± 1.2	15.3 ± 0.9	23.4 ± 2.4	221 ± 33	48 ± 5
			(360–3,280)	(6.3–21.8)	(9.5–20.6)	(11.5–42.7)	(67–500)	(22–70)
		Norway spruce (11)	1,009 ± 127	19.1 ± 1.6	17.9 ± 1.2	30.9 ± 3.0	323 ± 44	51 ± 5
			(620–2,040)	(10.2–27.7)	(9.8–23.6)	(10.5–44.5)	(68–586)	(26–78)
		Scots pine (9)	1,002 ± 181	17.3 ± 2.6	15.5 ± 2.4	20.8 ± 3.2	197 ± 51	59 ± 10
			(420–2,020)	(6.6–29.7)	(5.4–26.9)	(5.3–40.1)	(21–533)	(21–98)

Note: Mean values ± S.E. (minimum – maximum values) are summarized in the table.

Tree above-ground litter was collected using 5 litter collectors placed randomly in each research site under uniform forest canopy during the period from October 2018 till December 2019 (covering one full calendar year). Tree above-ground litter included everything falling from trees (foliage, branches, twigs, bark, fruits, seeds, rest of fruiting, fines, frass, insects, lichen, moss, etc.) excluding large dimension branches which are not perceived by collectors. This fraction of large dimension branches was not collected by collectors and is accounted under dead wood pool (natural mortality) within the National GHG Inventory. Thus, double accounting of C input to soil is avoided. The litter collector design – the collecting area of individual traps – 0.42 m^2, solid funnel (0.7 m deep) with a bag of inert material (nylon fabric) with mesh size of 0.2 mm. Above-ground litter was collected monthly (Ukonmaanaho et al. 2016). After transporting the tree above-ground litter to the laboratory, dry matter was determined by drying samples at a temperature of 105 °C to complete desiccation. Total C and N concentration of the grounded litter samples (dried at a temperature of 70 °C) were determined by total combustion at 950 °C with elemental analyser Elementar EL Cube according to the LVS ISO (2006) and ISO (1998), respectively.

Carbon input with tree below-ground litter (modelling approach using the NFI data)

Neumann et al. (2019) compiled data from 454 plots across forests in Europe and 19 estimation models of fine root biomass and production. We chose this model to estimate fine root biomass, which requires stem biomass as input data (given in Equation 1).

$$\textit{Fine root biomass}\ (t\ ha^{-1}) = \textit{stem biomass}\ (t\ ha^{-1}) \cdot 0.02 \quad (1)$$

Subsequently, we multiplied the biomass value by fine root turnover rate (yr^{-1}) to obtain the value of annual tree below-ground litter input. Yuan and Chen (2010) reviewed fine root characteristics in boreal forest ecosystems, and we used the turnover rates for *Betula* (1.22 ± 0.56), *Picea* (0.84 ± 0.07) and *Pinus* (0.61 ± 0.17) species from their study. To calculate C input with fine roots, it was assumed that the C content in biomass is 48% for broadleaves and 51% for conifers (Lamlom and Savidge 2003, IPCC 2006).

Carbon input with ground vegetation litter (modeling approach using the NFI data)

We used the equations elaborated by Muukkonen and Mäkipää (2006). Ground vegetation biomass (kg ha^{-1}) was calculated for spruce, pine and birch forest stands and for different plant forms such as mosses, lichens, dwarf shrubs, herbs and grasses separately (Equations 2–11). The input variable is stand age (years).

Pine forest stands:

Above-ground biomass (y), dwarf shrubs:

$$\sqrt{y+0.5} = 16.68 + 0.129 \cdot \textit{stand age} - 0.0004 \cdot \textit{stand age}^2 \quad (2)$$

Above-ground biomass (y), herbs and grasses:

$$\sqrt{y+0.5} = 11.725 - 0.098 \cdot \textit{stand age} + 0.0002 \cdot \textit{stand age}^2 \quad (3)$$

Above-ground biomass (y), mosses:

$$\sqrt{y+0.5} = 27.329 + 0.138 \cdot stand\ age - 0.0005 \cdot stand\ age^2 \quad (4)$$

Above-ground biomass (y), lichens:

$$\sqrt{y+0.5} = 7.975 - 0.0002 \cdot stand\ age^2 \quad (5)$$

Spruce forest stands:

Above-ground biomass (y), dwarf-shrubs:

$$\sqrt{y+0.5} = 10.375 - 0.033 \cdot stand\ age + 0.001 \cdot stand\ age^2 - 0.000004 \cdot stand\ age^3 \quad (6)$$

Above-ground biomass, herbs and grasses:

$$\sqrt{y+0.5} = 15.058 - 0.113 \cdot stand\ age + 0.0003 \cdot stand\ age^2 \quad (7)$$

Above-ground biomass (y), mosses:

$$\sqrt{y+0.5} = 19.282 + 0.164 \cdot stand\ age - 0.000001 \cdot stand\ age^3 \quad (8)$$

Broad-leaved forest stands:

Above-ground biomass (y), dwarf-shrubs:

$$\sqrt{y+0.5} = 7.102 + 0.0004 \cdot stand\ age^2 \quad (9)$$

Above-ground biomass (y), herbs and grasses:

$$\sqrt{y+0.5} = 20.58 - 0.423 \cdot stand\ age + 0.004 \cdot stand\ age^2 - 0.00002 \cdot stand\ age^3 \quad (10)$$

Above-ground biomass (y), mosses:

$$\sqrt{y+0.5} = 13.555 - 0.056 \cdot stand\ age \quad (11)$$

To calculate above-ground vegetation litter, the obtained values were multiplied by the turnover rates of the respective plant forms – 0.25 for dwarf-shrubs, 1 for herbs and grasses, 0.33 for mosses and 0.1 for lichens (Muukkonen 2006). It was assumed that the proportion of the ground vegetation biomass located in the below-ground parts is 70% of the total biomass (Mälkönen 1974, Havas and Kubin 1983, Palviainen et al. 2005). To calculate C input with ground vegetation litter, it was assumed that the C fraction in biomass is 0.475 (Magnussen and Reed 2015).

The National Forest Inventory data used for calculations

Characteristic parameters of the forest stands with drained organic soils (Table 2) provided by the 3rd cycle of the National Forest Inventory (NFI) were used to model tree fine root biomass according to the Equation 1 and above-ground biomass of ground vegetation according to the Equations 2–11.

Applied soil emission factors

Table 3 summarizes the applied GHG EFs for forests with drained organic soils in the hemiboreal region. The range of basal areas in the forests, where the applied heterotrophic respiration values (GHG EFs) were measured, is from 12.8 to 28.3 m^2 ha^{-1} (mean 20.5 m^2 ha^{-1}) for Scots pine stands and from 14.9 to 28.1 m^2 ha^{-1} (mean 20.7 m^2 ha^{-1}) for silver birch stands (Lazdiņš and Lupiķis 2019). Forests, where the applied GHG EFs were measured, correspond to *Myrtillosa turf. mel.* (relatively high soil fertility) forest site type (Lazdiņš and Lupiķis 2019).

Net GHG emissions were calculated as a sum of GHG emission from soil and total C input to soil. Emissions are usually expressed with a positive sign, but removals including C input to soil – with a negative sign. Respectively, negative net GHG emissions mean that the system is a net sink contributing to reduction of GHG emissions, and if net GHG emissions have a positive sign – the system is a net source of GHG emissions contributing to increase of GHGs in atmosphere (IPCC 2006).

Table 2. Average characteristic parameters of the forest stands with drained organic soils in Latvia (NFI, 3rd cycle)

Parameter	Value	Dominant tree species		
		Scots pine	Norway spruce	Silver birch
Number of plots	number	349	242	503
Age of dominant tree species, years	average ± S.E.	79 ± 2	51 ± 2	41 ± 1
	range (min…max)	1–221	1–195	1–119
Total basal area, m^2 ha^{-1}	average ± S.E.	26.7 ± 1.5	22.8 ± 1.5	18.0 ± 0.6
	range (min…max)	0.0079–90.7	0.0079–130.8	0.0028–85.4
Stem biomass, t ha^{-1}	average ± S.E.	210.7 ± 7.2	167.2 ± 8.5	145.3 ± 5.8
	range (min…max)	0.008–760.8	0.011–787.7	0.002–924.2

Table 3. Applied GHG emission factors for the forests with drained organic soils in the hemiboreal zone

Dominant tree species	CO_2-C *, t ha^{-1} yr^{-1}	CH_4-C, kg ha^{-1} yr^{-1}	N_2O-N, kg ha^{-1} yr^{-1}	CH_4 from drainage ditch ***, kg ha^{-1} yr^{-1}	Total GHG, t CO_2-C eq. ha^{-1} yr^{-1}
Silver birch	5.60	22.39	0.62	217	5.91
Norway spruce	5.25	–1.39 **	–0.05 **	217	5.27
Scots pine	5.25	–1.39	–0.05	217	5.27
Data source	Lazdiņš and Lupiķis 2019	Lazdiņš and Lupiķis 2019	Lazdiņš and Lupiķis 2019	IPCC 2014	Calculated

Note: * Soil heterotrophic respiration; ** Emission factor estimated for Scots pine dominated stands (Lazdiņš and Lupiķis 2019) was used; *** A fraction of the total area of drained organic soil which is occupied by ditches is 2.5% (IPCC 2014).

Data analysis

Data processing and all statistical analyses were performed in the R environment (R Core Team 2017). Statistical differences between average values were analysed with the pairwise comparison using *t* test with pooled SD (function *pairwise.t.test()*). Correlations (including their significance) between biomass of tree above-ground litter and characteristics of the forest stands were tested using Pearson's product-moment correlation test (function *cor.test()*). We considered relationships significant if *p* values were lower than 0.05. To gain a better understanding of relationships between annually produced biomass of tree above-ground litter and characteristics of forest stand, a linear equation models were constructed. For all analyses, a 95% confidence level was used.

Results

Above-ground litter of tree

The research site average annually produced biomass of tree above-ground total litter in the forests with drained organic soils was within the range from 1.08 ± 0.16 t ha^{-1} yr^{-1} in the Scots pine dominated stand which is the youngest forest stand included in the study and characterized with the lowest stem biomass parameters to 7.26 ± 0.39 t ha^{-1} yr^{-1} in the Norway spruce dominated stand with relatively high stem biomass parameters. Average annually produced biomass of tree above-ground litter in the research sites was 3.77 ± 0.23 t ha^{-1} yr^{-1}.

Table 4 summarizes statistical data on the relationships characterizing a dependence of annually produced biomass of tree above-ground litter on characteristics of the forest stands with drained organic soils. For the silver birch and Norway spruce stands strong and statistically significant correlations ($r > 0.7, p < 0.05$) were found between annually produced biomass of tree above-ground litter and forest stand characteristics such as average height, basal area and stock which in turn correlate with each other. For the Scots pine stands moderately strong, but statistically insignificant correlations ($0.5 < r < 0.7, p > 0.05$) between annually produced biomass of tree above-ground litter and average diameter and basal area of the forest stands were revealed.

Annually produced biomass of tree above-ground litter was best described by stand basal area as the most significantly influencing factor (independent variable) of nonlinear regressions. The best models for annual production of tree above-ground litter (polynomial regression for the silver birch stands and power regression for the Norway spruce and Scots pine dominated stands) are shown in Figure 2.

Average total C and N concentration, as well as average C/N ratio of tree above-ground litter are shown in Figure 3. In general, statistically significantly higher ($p < 0.001$) total C concentration was found in Scots pine (540.7 ± 2.6 g kg^{-1}) and silver birch (537.9 ± 2.1 g kg^{-1}) above-ground litter if compared to Norway spruce (521.2 ± 2.1 g kg^{-1}) above-ground litter. Total N concentration in Scots pine above-ground litter (8.0 ± 0.3 g kg^{-1}) was significantly smaller if compared to Norway spruce (14.2 ± 0.4 g kg^{-1}) and silver birch (14.2 ± 0.2 g kg^{-1}) above-ground litter. Consequently, significantly higher ($p < 0.001$) C/N ratio was found in Scots pine litter (70.6 ± 2.1) if compared to Norway spruce (37.9 ± 0.9) and silver birch (38.7 ± 0.8) above-ground litter (Figure 3).

The comparison of calculated total C and N annual input with tree above-ground litter between stands with different dominant tree species in the forests with drained

Table 4. Statistical data (correlation coefficients *r*, *p*-values, equations and adjusted R^2 of linear regressions) on the relationships characterizing dependence of annually produced biomass of tree above-ground litter on characteristics of the forest stand with drained organic soils

Tree species	Independent variable	Pearson's correlation		Linear regression	
		r	*p*	equation	adjusted R^2
Scots pine	Number of trees per hectare, count ha^{-1}	–0.37	0.330	$y = -0.00061x + 3.61$	0.012
	Diameter, cm	**0.50**	0.175	$y = 0.056x + 2.03$	0.14
	Height, m	0.42	0.258	$y = 0.054x + 2.17$	0.061
	Basal area, m^2 ha^{-1}	**0.56**	0.120	$y = 0.051x + 1.93$	0.21
	Stock, m^3 ha^{-1}	0.40	0.282	$y = 0.0024x + 2.53$	0.043
	Stand age, years	0.16	0.674	$y = 0.050x + 2.68$	–0.11
Silver birch	Number of trees per hectare, count ha^{-1}	–0.29	0.270	$y = -0.00046x + 4.76$	0.021
	Diameter, cm	0.46	0.075	$y = 0.12x + 2.45$	0.15
	Height, m	**0.71**	**0.002**	$y = 0.24x + 0.31$	0.47
	Basal area, m^2 ha^{-1}	**0.75**	**< 0.001**	$y = 0.09x + 1.88$	0.52
	Stock, m^3 ha^{-1}	**0.73**	**0.001**	$y = 0.0068x + 2.55$	0.50
	Age, years	0.46	0.073	$y = 0.32x + 2.41$	0.16
Norway spruce	Number of trees per hectare, count ha^{-1}	–0.13	0.698	$y = -00045x + 5.83$	–0.092
	Diameter, cm	**0.59**	0.055	$y = 0.17x + 2.20$	0.28
	Height, m	**0.73**	**0.011**	$y = 0.26x + 0.72$	0.48
	Basal area, m^2 ha^{-1}	**0.82**	**0.002**	$y = 0.12x + 1.65$	0.65
	Stock, m^3 ha^{-1}	**0.78**	**0.005**	$y = 0.0077x + 2.87$	0.56
	Age, years	0.10	0.771	$y = 0.096x + 4.84$	–0.10

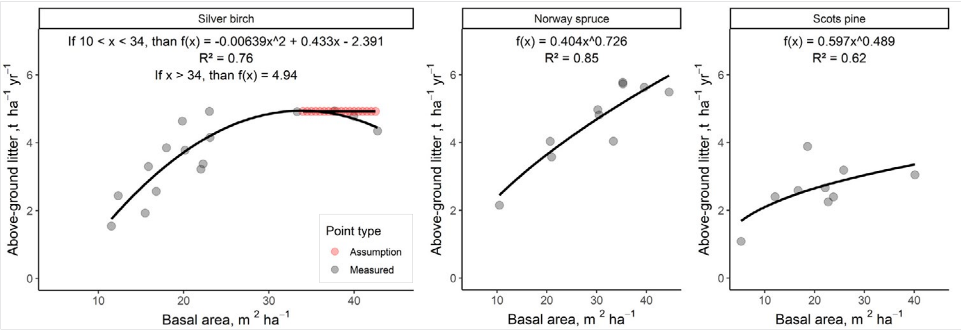

Figure 2. Nonlinear regressions describing dependence of annually produced biomass of tree above-ground litter on basal area in the forests with drained organic soils

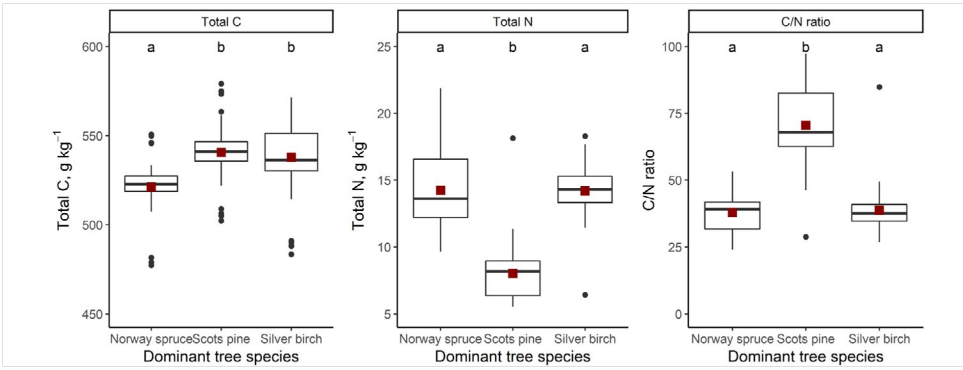

Figure 3. Total C and N concentrations and C/N ratio in tree above-ground litter in forests with drained organic soils based on field measurements

Note: In the boxplots, the median is shown by the bold line, the mean is shown by the dark red square, the box corresponds to the lower and upper quartiles, whiskers show the minimal and maximal values (within 150% of the interquartile range from the median) and black dots represent outliers of the datasets. Characters *a* and *b* label statistically significant differences ($p < 0.05$, $\alpha = 0.05$) in average values between stands with different dominant tree species.

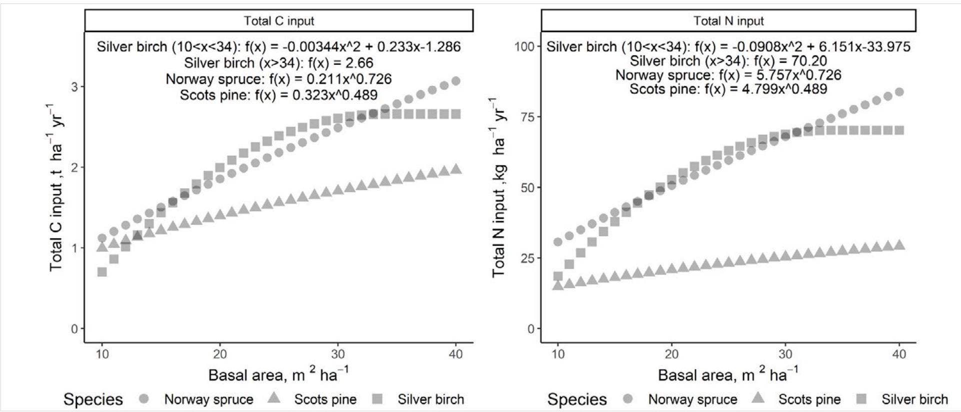

Figure 4. Calculated total C and N input through above-ground tree litter in stands characterized with basal area in the range from 10 to 40 m^2 ha^{-1} in the forests with drained organic soils

organic soils is shown in Figure 4. We calculated the input of total C and N by applying equations of nonlinear regressions of annual input of above-ground litter biomass depending on stand basal area (Figure 2) and average C and N concentrations in litter of different tree species (Figure 3). In the stands with a range of basal area from 10 to 40 m^2 ha^{-1}, the highest total C and N annual input was estimated in the Norway spruce dominated stands with a basal area of 40 m^2 ha^{-1}, but the smallest total C and N annual input was estimated in the silver birch and Scots pine dominated stands, respectively, with a basal area of 10 m^2 ha^{-1} (Figure 4).

Modelled carbon input through tree below-ground litter

Modelled C annual input through tree below-ground litter in the forests with drained organic soils based on stem biomass data provided by the NFI (3rd cycle) is shown in Figure 5. In the Scots pine dominated stands, weighted average C annual input through below-ground tree litter that takes into account the distribution of forest stands according to the NFI data was 1.31 ± 0.05 t ha^{-1} yr^{-1}. In the Norway spruce dominated stands it was 1.43 ± 0.07 t ha^{-1} yr^{-1}, but in the silver birch stands – 1.70 ± 0.07 t ha^{-1} yr^{-1}, furthermore, differences in average values between stands with different dominant tree species were statistically significant ($p < 0.003$). The highest average C annual input through below-ground tree litter (3.52 ± 0.97 t ha^{-1} yr^{-1}) was estimated in the silver birch dominated stands at the age of > 91 years, but the lowest C input was estimated in the young stands of silver birch up to 10-years age (0.07 ± 0.02 t ha^{-1} yr^{-1}).

Modelled carbon input through ground vegetation litter

The modelled total C annual input through above-ground and below-ground litter of ground vegetation (dwarf shrubs, herbs, grasses, mosses and lichens) in the forests with drained organic soils is shown in Figure 6. The modelled total C annual input through above- and below-ground litter of ground vegetation ranges up to 1.55 ± 0.18 t ha^{-1} yr^{-1} in the Norway spruce dominated stands with the age of > 140 years. The weighted average annual C input through above-ground and below-ground litter of ground vegetation that takes into account the distribution of forest stands according to the NFI data in the Scots pine dominated stands was 0.91 ± 0.01 t ha^{-1} yr^{-1}, in the Norway spruce dominated stands – 0.65 ± 0.01 kg ha^{-1} yr^{-1}, but in silver birch stands – 0.27 ± 0.01 t ha^{-1} yr^{-1}, furthermore, differences in average values between stands with different dominant tree species were statistically significant ($p < 0.001$).

In the Norway spruce and Scots pine dominated stands, mosses produce the largest share of the C input through above-ground litter of ground vegetation (61 and 68% of total C input, respectively). The second largest share of the C input through above-ground litter of ground vegetation is formed by herbs and grasses in the Norway spruce dominated stands (29% of total C input) and dwarf shrubs in the Scots pine dominated stands (25% of total C input). In the silver birch dominated stands, the largest share of C input through above-ground litter of ground vegetation is formed by herbs and grasses (52% of total C input), but the second largest share of C input through above-ground litter of ground vegetation is produced by mosses (32% of total C input).

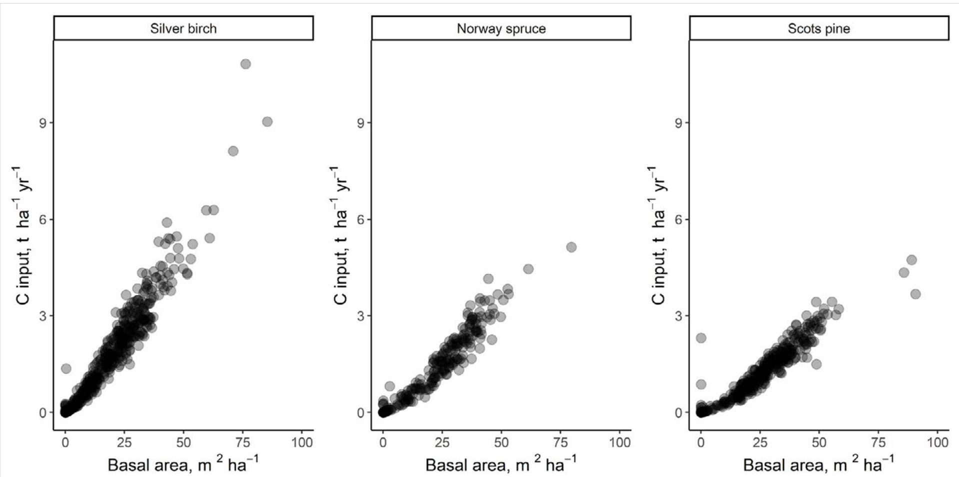

Figure 5. Modelled carbon input through below-ground tree litter in the forests with drained organic soils based on stem biomass data provided by the National Forest Inventory (3rd cycle)

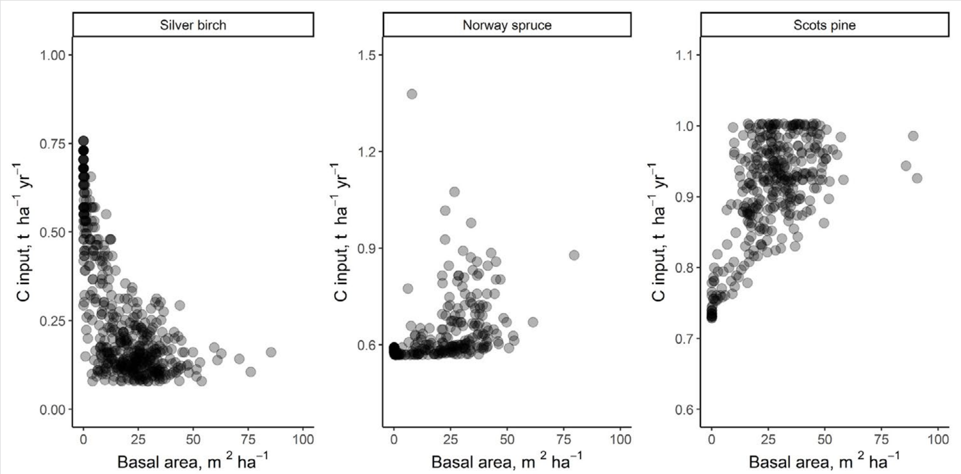

Figure 6. Modelled carbon input through above-ground and below-ground litter of ground vegetation (dwarf shrubs, herbs, grasses, mosses and lichens) in the forests with drained organic soils based on stand age distribution data provided by the National Forest Inventory (3rd cycle)

Net GHG emissions from soil

Net GHG emissions from soils in the forests with drained organic soils were calculated for stands characterized with basal area in the range between 10 and 40 m^2 ha^{-1} (Figure 7). In the forest stands within this basal area range, the calculated individual net GHG emissions from soils ranged from 4.30 t CO_2-C ha^{-1} yr^{-1} to –2.15 t CO_2-C ha^{-1} yr^{-1} (both minimum and maximum value detected in the silver birch dominated stands). Weighted average net GHG emissions that takes into account the distribution of forest stands according to the NFI data were 1.54 ± 0.05 t CO_2-C ha^{-1} yr^{-1} in the Scots pine dominated stands, 0.70 ± 0.10 t CO_2-C ha^{-1} yr^{-1} in the Norway spruce dominated stands and 1.47 ± 0.08 t CO_2-C ha^{-1} yr^{-1} in the silver birch dominated stands.

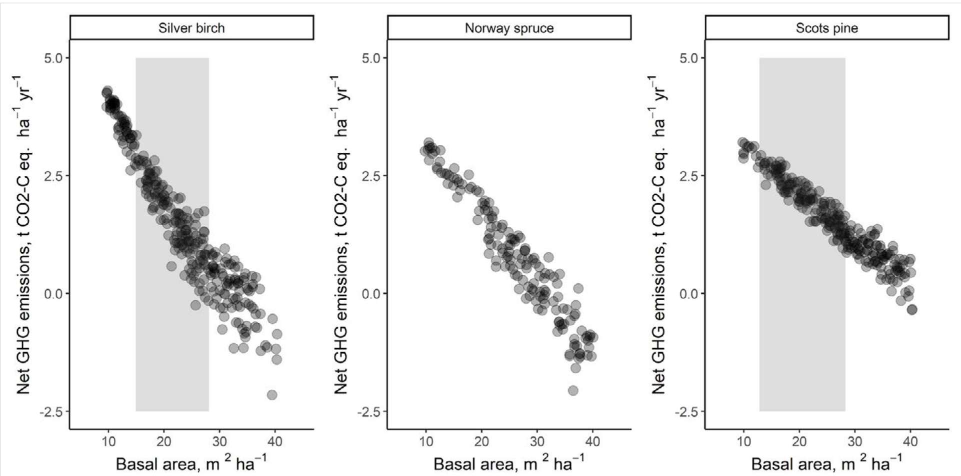

Figure 7. Net GHG emissions from soil in stands characterized with basal area in the range from 10 to 40 m^2 ha^{-1} in the forests with drained organic soils

Note: A grey area indicates the range of basal areas in the forests where the applied heterotrophic respiration (GHG emission factors, Table 3) values were measured.

In general, drained organic soils in the silver birch and Norway spruce stands with basal area in the range from 10 to 32 and 31 m^2 ha^{-1}, respectively, and in the Scots pine stands with basal area in the range from 10 to 40 m^2 ha^{-1} were source of net GHG emissions. But drained organic soils in the silver birch stands with basal area in the range from 32 to 40 m^2 ha^{-1} and in the Norway spruce stands with basal area in the range from 31 to 40 m^2 ha^{-1} were sink of net GHG emissions. In stands within these basal area ranges (32–40 m^2 ha^{-1} for silver birch and 31–40 m^2 ha^{-1} for Norway spruce stands), weighted average net GHG emissions were –0.29 ± 0.09 t CO_2-C ha^{-1} yr^{-1} in the silver birch stands and –0.61 ± 0.09 t CO_2-C ha^{-1} yr^{-1} in the Norway spruce. Underestimation or overestimation of total net GHG emissions might have occurred in stands with basal area ranges not covered by estimates of soil heterotrophic respiration (uncoloured area in Figure 7).

Discussion

Biomass of tree above-ground litter

Litter production is a significant process in the global C and nutrient cycles of terrestrial ecosystems (Liu et al. 2004, Feng et al. 2019). Tree species and climate are key drivers for litter production; thus, litter production rates are usually estimated by biogeoclimatic zones using equations including climatic parameters (e.g. Berg and Meentemeyer 2001, Liu et al. 2004) and information on biomass abundance (e.g. Neumann et al. 2018) as predictors. For instance, Berg and Meentemeyer (2001) developed regressions for European coniferous forests against a set of climatic parameters, and the best simple relationships were obtained with annual actual evapotranspiration and other parameters including temperature, whereas, for example, precipitation gave lower *r* values. Based on the review of original publications over litter production in Eurasian forests Liu et al. (2003) calculated that average total litter production rate in boreal forests is 2.61 ± 1.08 t ha^{-1} yr^{-1} with a range from 0.27 to 5.08 t ha^{-1} yr^{-1}. They developed regression model that uses annual mean temperature and annual precipitation as independent variables (Liu et al. 2004). Similarly, as Berg and Meentemeyer (2001), Liu et al. (2004) also concluded that annual mean temperature has a greater effect on litter production compared to annual precipitation across the Eurasian forests. The mean values from the data provided by the ICP Forests Level II network covering the full geographical range of European forests (Neumann et al. 2018) are higher than calculated by Liu et al. (2004). Neumann et al. (2018) calculated that average annual litter production rate for northern Europe (Fennoscandia and Baltic states, mainly boreal forests) is 3.22 ± 2.01 t ha^{-1} yr^{-1} for conifers and 2.76 ± 1.27 t ha^{-1} yr^{-1} for broadleaves. Further they concluded that the best developed regression model for total litter production uses temperature, precipitation and biomass abundance (stand density and leaf area index) as independent variables (Neumann et al. 2018). We developed country-specific regression model for total tree above-ground litter production in the stands with drained organic soils using stand basal area as an independent variable; climatic parameters were omitted from the models due to narrow coverage of climate transect by the research sites. Field observations showed tree above-ground litter production rate in the forests with drained organic soils in the range from 1.08 ± 0.16 to 7.26 ± 0.39 t ha^{-1} yr^{-1} depending on dominant tree species and forest stand biomass parameters.

Most of the regional evaluations of litter production rates carried out so far do not differentiate forests with organic soils, although forest stands with organic soils may structurally differ from stands on mineral soils (Laiho et al. 2003, Laiho et al. 2008). According to the 3rd cycle of the NFI data, 73% of the Scots pine stands and 65% of the Norway spruce stands with drained organic soils in Latvia correspond to the basal area range from 10 to 40 m^2 ha^{-1} and average litter production (biomass) rate in these stands is 2.90 ± 0.03 and 4.33 ± 0.08 t ha^{-1} yr^{-1}, respectively (calculated based on the regression models developed within the study). Most of the silver birch stands with drained organic soils (60%) correspond to the basal area range from 10 to 40 m^2 ha^{-1} as well, and average litter production (biomass) rate in these stands is 3.86 ± 0.06 t ha^{-1} yr^{-1}. These calculated values of average litter production rate in the forests with drained organic soils in Latvia are significantly higher than those calculated using, for instance, the regression models developed by Liu et al. (2004) which use annual mean temperature and annual precipitation as independent variables (1.48 t ha^{-1} yr^{-1} for broadleaves and 1.88 t ha^{-1} yr^{-1} for conifers if annual mean temperature is 8.1 °C and annual precipitation is 732 mm).

Carbon input through tree above-ground litter

Mostly, C content in conifers is higher than in broadleaves due to higher lignin content in coniferous wood (Lamlom and Savidge 2003), but exceptions are observed in northern Europe (Neumann et al. 2018) which was also confirmed by our results. The default IPCC (2006) C content for temperate and boreal regions in above-ground forest biomass of 48% for broadleaves and 51% for conifers (Lamlom and Savidge 2003, IPCC 2006) provides estimates, which are about 12% lower for broadleaves and about 4% lower for conifers than C content estimates in litter determined within our study. Thus, based on our results, we can support the use of both tree species- and region-specific C content values within estimations of C flows through litter production since C content in litter differs significantly between tree species and biogeoclimatic zones.

A high C/N ratio may indicate slower decomposition rates due to high lignin/N ratios that retard the decomposition processes (Berg et al. 2000). Furthermore, litters rich in N (with a low C/N ratio) not only decompose faster, but also increase the decomposer activity (C-use efficiency), resulting in C transportation, incorporation and ultimately stabilization into the deeper soil matrix (Zhou et al. 2019). Results of our study indicated that the Norway spruce and silver birch stands produce litter with a significantly higher total N content and lower C/N ratio if compared to Scots pine litter, which theoretically can promote higher SOC accumulation rate in the Norway spruce and silver birch stands.

Neumann et al. (2018) estimated that the average C input through total tree above-ground litter in the forests of northern Europe is 1.7 ± 1.1 t C ha^{-1} yr^{-1} for conifer stands and 1.5 ± 0.7 t C ha^{-1} yr^{-1} for broadleaved stands. Our average estimates of C input for conifer and silver birch stands characterized with basal area in the range from 10 to 40 m^2 ha^{-1} (1.82 ± 0.02 t C ha^{-1} yr^{-1} and 2.07 ± 0.03 t C ha^{-1} yr^{-1}, respectively) are within the range of their estimates. Our average estimates for C input through tree above-ground litter in the stands with basal area in the range from 10 to 40 m^2 ha^{-1} were about 1.5 times higher for conifers, comparing with the input from tree below-ground litter. For silver birch, average estimates for C input through tree above-ground litter were quite like C input through tree below-ground litter (average difference in C input between tree above- and below-ground litter was –0.13 t C ha^{-1} yr^{-1}).

Carbon input through tree below-ground litter

Our estimates fall within the range of the results of other studies carried out in the boreal region and show a tendency for below-ground litter to increase with increasing stand basal area. According to our estimates, the below-ground litter input also tends to increase along with stand age. The highest litter input was observed in the silver birch stand (with the age of > 50 years). Underestimation or overestimation may have occurred because there were no LAI or foliage biomass data available and estimation models that require these parameters as input variables offer the most accurate results.

According to a study carried out in southern Sweden, the estimated fine root litter input was the highest in spruce stands (1.3 t C ha^{-1} yr^{-1}) followed by pine (1.06 t C ha^{-1} yr^{-1}) and birch (0.77 t C m^{-2} yr^{-1}) stands (Hansson et al. 2011). Also, in a study carried out by Ågren et al. (2007) using data from the Swedish forest inventories it was concluded that fine root turnover influenced C sequestration in spruce forests more significantly than in pine forests. Other aspects that should be considered along with the dominant tree species are nutrient availability and soil temperature. In a nutrient manipulation experiment carried out in a Norway spruce stand in northern Sweden Leppälammi-Kujansuu et al. (2014) found that the fine root lifespan was significantly shorter in warmer and more nutrient-rich soil and the litter input increased. This aspect should be considered in the context of global warming and increasing soil temperatures. In the control treatment the C input with fine root litter was 0.51 t C ha^{-1} yr^{-1}. Fertilization or warming alone increased the amount of below-ground litter production (1.476 and 1.45 t C ha^{-1} yr^{-1}, respectively), whereas both treatments simultaneously increased the fine root litter input 4.4-fold (2.246 t ha^{-1} yr^{-1}). Along with improving nutrient availability trees invest less in nutrient acquisition and more resources are invested in above-ground biomass production (Vanninen and Mäkelä 1999, Iivonen et al. 2006).

Carbon input through ground vegetation litter

Our results show a weak positive association between C input with ground vegetation litter and stand basal area in the conifer stands. The trend is more pronounced in pine stands. In the silver birch stands there is a logarithmic trend for C input to decrease along with increasing stand basal area. C input varies between stands of each dominant tree species because of differences in ground vegetation composition and abundance of certain species. Studies pursued by other researchers show that ground vegetation composition is strongly linked to stand age and differences in canopy cover (MacLean and Wein, 2011, Bäcklund et al. 2015, Majasalmi and Rautiainen, 2020). The lower ground vegetation litter production in the birch stands could be explained by poorer light availability resulting from denser canopy. Also the difference between pine and spruce-dominated stands may have occurred due to lower light levels that reach the understory of the latter. The ground vegetation abundance in spruce stands is generally lower than in pine and it decreases rapidly with increasing stem density (Hedwall et al. 2013, Bäcklund et al. 2015, Tonteri et al. 2016). In pine stands light is rarely the limiting factor, and ground vegetation is primarily influenced by competition (Tonteri et al. 1990). Other studies conducted in boreal forests show that the field layer (dwarf shrubs, herbs and grasses) of ground vegetation declines with increasing stand basal area, whereas no conclusions can be drawn regarding the C stock in the moss layer (Muukkonen and Mäkipää 2006, Hansson et al. 2011). Some studies show that the cover of grasses tends to decline with reduced light availability under a dense canopy, whereas the cover of dwarf shrubs tends to initially increase, then decrease and eventually bryophytes dominate (Hedwall et al. 2013, Felton et al., 2020). Overstory effects also depend on interspecific competition and soil condition (Kuusipalo 1985). Our results confirm that biomass of herbs declines with increasing stand age and that dwarf shrub biomass is slightly increasing. While the decline of vascular plants has a considerable impact on ground vegetation litter production and annual C input because of fast turnover rates, the increase in dwarf shrub biomass has a negligible impact. The biomass of mosses is slightly decreasing along with increase of stand basal area.

Kristensen et al. (2015) studied above- and below-ground C pools in boreal forests using LiDAR and found that 1.64–3.31 t C ha^{-1} is in the ground vegetation compartment. Lehtonen et al. (2016) estimated that the mean C input from ground vegetation is approximately 0.473 and 0.863 t ha^{-1} C for the southern and northern parts of Finland, respectively, which correspond to our results obtained in the spruce stands, but are lower than our estimates for the pine stands and higher than those for the birch stands. Hansson et al. (2011) estimated that litter production by shrubs and ground vegetation was higher in birch (0.84 t C ha^{-1} yr^{-1}) and pine (0.71 t C ha^{-1} yr^{-1}) than in spruce stands (0.24 t C ha^{-1} yr^{-1}), however the values are difficult to compare with our results, when mosses are excluded.

As it was indicated in the study conducted by Muukkonen and Mäkipää (2006), equations including site attributes like latitude, longitude, elevation, temperature sum, nutrient level, stem volume, number of trees per 1 ha, basal area and stand age – as input variables offer more accurate estimates than the equations with stand age alone as an input variable, however these equations are country-specific and can be applied only in Finland. The equations used in our study were originally developed for upland forest stands, which could be another reason for inaccuracies in our estimations. Additionally, ground vegetation is a variable component of the forest ecosystem, therefore it cannot be predicted with conventional site attributes only. Site disturbances as well as interspecies' relationships can significantly influence the species composition and biomass of the ground vegetation.

To obtain more accurate estimates of C input with ground vegetation, it is required to investigate which site attributes can be used to predict ground vegetation biomass and to develop country-specific ground vegetation biomass equations for peatland forests in Latvia.

Net GHG emissions from soil

Litter production is one of the most important ecological processes in forest ecosystems, influencing the C and nutrient transfer from vegetation to the soil (Liu et al. 2004), but organic matter stored in soil may be significantly affected by different land management practices or changes in the predominant climatic patterns (Laiho et al. 2008). Any land management-mediated changes in SOC stock and GHG emissions from soils need to be estimated and reported within the National GHG Inventories. Boreal and temperate forests with drained organic soils may act either as a sink (e.g. Minkkinen and Laine 1998, Ojanen et al. 2013, Lupiķis and Lazdiņš 2017, Ojanen et al. 2019) or a source of C (e.g. Simola et al. 2012, Pitkänen et al. 2013, Hommeltenberg et al. 2014) depending on the case, but the combinations of factors controlling this variation are still insufficiently understood (Laiho et al. 2008).

Results of our study obtained by combining field observations (C input through tree above-ground litter), modelling approach (C input through tree below-ground litter and litter of ground vegetation) and NFI data on characteristics of forest stands with drained organic soils showed that drained organic soils in the silver birch and Norway spruce stands with basal area in the range from 10 to 32 and 31 m^2 ha^{-1}, respectively, and in the Scots pine stands with basal area in the range from 10 to 40 m^2 ha^{-1} were a source of net GHG emissions. At the same time, drained organic soils in the silver birch stands with basal area in the range from 32 to 40 m^2 ha^{-1} and in the Norway spruce stands with basal area in the range from 31 to 40 m^2 ha^{-1} were a sink of net GHG emissions (-0.29 ± 0.09 t CO_2-C ha^{-1} yr^{-1} in the silver birch stands and -0.61 ± 0.09 t CO_2-C ha^{-1} yr^{-1} in the Norway spruce stands). Furthermore, it should be noted that C input through natural mortality of tree biomass (including large dimension branches and parts of stumps and roots), which is a significant source of C to soil, was not included in the assessment of net GHG emissions from the system. According to the Latvia's National Inventory Report, the weighted average natural mortality in 2017 was 2.01 m^3 ha^{-1} yr^{-1} in Latvia, and it corresponds to 0.72 t C ha^{-1} yr^{-1}. Thus, net GHG emissions from drained organic soils calculated within the study could be overestimated. Considering that C input through natural mortality was not included, the results obtained within this study approach the results reported by Lupiķis and Lazdiņš (2017) who concluded that in the hemiboreal vegetation zone drainage of organic soils is not always causing C storage reduction.

Although soil heterotrophic respiration increases with increasing litter production (soil fertility impact) (e.g., Ojanen et al. 2013), we used the constant GHG EFs for the forests with drained organic soils without division into fertile and poor sites (Table 3) due to lack of more stratified GHG emission data corresponding to the hemiboreal zone. This could underestimate or overestimate total calculated net GHG emissions in stands with basal area range not covered by estimates of soil heterotrophic respiration. To obtain more accurate estimates of net GHG emissions from the forest stands with drained organic soils, it is required to include dynamic data of soil heterotrophic respiration depending on stand fertility and dynamics of litter production in calculations.

Conclusions

Drained organic soils in silver birch, Scots pine and Norway spruce dominated stands in hemiboreal conditions may act either as a sink or a source of net GHG emissions depending mostly on characteristics of the stand (both stand age, growing stock and basal area); furthermore, the variation in calculated net GHG emissions was relatively large. It highlights the need to include the stratified EFs for drained organic soils depending on dominant tree species and stand characteristics in the National GHG Inventories.

It is necessary to conduct further research to get a better understanding of C flows in drained organic soils covering forest stands with a wider range of basal area stratified by soil fertility and GHG fluxes in forests with naturally wet organic soils. It would contribute not only to more accurate estimates of net GHG emissions for the National GHG Inventories, but also to the development of a more sustainable management of forests with organic soils.

Acknowledgements

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