A new method for estimating carbon dioxide emissions from drained peatland forest soils for the greenhouse gas inventory of Finland

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Abstract Reporting the greenhouse gas (GHG) emissions from the LULUCF sector in the GHG inventory requires sound methods for estimating both the inputs and outputs of carbon (C) in managed ecosystems. Soil CO_2 balance of forests
15   consists of the CO_2 released from decomposing soil organic matter (SOM) and the C entering the soil through aboveground and belowground plant litter input. Peatlands drained for forestry release soil C as CO_2 because the drainage deepens the oxic peat layer prone to SOM decomposition. IPCC Guidelines provide default CO_2 emission factors for different climatic zones and the defaults or locally adapted static emission factors are commonly in use in GHG inventory reporting for drained peatlands. In this paper, we describe a new dynamic method to estimate the CO_2 balance of drained peatland forest soils in
20   Finland. Contrary to static emission factors, the annual CO_2 release from soil is in our method estimated using empirical regression models driven by time series of tree basal area (BA), derived from the national forest inventories in Finland, time series of air temperature and the drained peatland forest site type. Aboveground and belowground litter input is also estimated using empirical models with newly acquired turnover rates for tree fine roots and BA as a dynamic driver. All major components of litter input from ground vegetation and live, harvested and naturally died trees are included. Our
25   method produces an increasing trend of emissions from 1.4 to 7.9 Mt CO_2 for drained peatland forest soils in Finland for the period 1990–2021, with a statistically significant difference between years 1990 and 2021. Across the period 1990–2021, annual emissions are on average 3.4 Mt and -0.3 Mt in southern and northern parts of Finland, respectively. When combined with data of the CO_2 sink created by trees, it appears that in 2021 drained peatland forest ecosystems were a source of 2.3 Mt CO_2 in southern Finland and a sink of 2.5 Mt CO_2 in northern Finland. We compare the emissions produced by the new
30   method with those produced by the old GHGI method of Finland and discuss the strengths and vulnerabilities of our method in comparison to static emission factors.

1 Introduction

Parties and signatories of the Paris Agreement and the United Nations Framework Convention on Climate Change (UNFCCC) are committed to reporting their anthropogenic greenhouse gas (GHG) emissions and removals by sources and sinks, respectively. In peatlands, drainage increases aerobic decomposition of soil organic matter (SOM) and leads to substantial carbon dioxide (CO_2) emissions, globally accounting for 6 % of total CO_2 emissions (Joosten 2010). These emissions are reported under Land use, land-use change and forestry (LULUCF) sector of the GHG inventory.

Almost one third, or 8.8 Mha (million hectares) of land area in Finland is covered by peatlands. This area also includes ca. 1 Mha thin-peated sites (< 30 cm peat layer). Of the total peatland area, 4.7 Mha is drained for forestry (Korhonen et al. 2021), 0.34 Mha for agriculture and 0.11 Mha for peat extraction (Statistics Finland 2022). Peatland forests have mainly been drained after World War II, with a peak in activity in 1960s and 1970s (Päivänen and Hånell, 2012). This paper addresses soil CO_2 emissions from those 4.3 Mha of all drained peatland forests in Finland that fulfil the FAO definition of forest land. Soil CO_2 emissions from undrained peatland forests are excluded as the emissions there are in the GHG inventory assumed to be in equilibrium. Emissions of CO_2 from drained peatland forest soils were first globally estimated by IPCC (2006). In Finland, the emissions, or soil C balance, have been quantified by Minkkinen et al. (2002) and Ojanen et al. (2014). Drained peatland forest soils are a significant source of CO_2 emissions in Finland: in 2020, they were a source of 3.8 Mt CO_2, while the LULUCF sector in total was a sink of 17.3 Mt CO_2 eq. (Statistics Finland 2022).

Countries may use static IPCC default emission factors for reporting soil CO_2 emissions from their drained peatland forest soils. Country-specific data and improvement of GHG inventory methods beyond the IPCC default emission factors are relevant especially for countries that have large areas of drained peatlands. Some countries have indeed opted to use country-specific emission factors; these can be based on the subsidence, i.e. fall of the surface of the organic soil, as in The Netherlands (National Institute for Public Health and the Environment 2021), CO_2 flux measurements as in Ireland (Duffy et a. 2021), or soil C balance modelling as in the UK (Brown et al. 2021). Weakness of static emission factors is that they cannot follow temporal changes, if such occur, in variables that drive the soil CO_2 balance. For instance, net CO_2 emissions from peatland forest soils may gradually evolve after drainage due to a successional transition of vegetation and changes in peat decomposition and hydrology (Sarkkola et al. 2010, Straková et al. 2012). In addition, increasing global temperatures likely increase SOM decomposition rates and thus soil CO_2 emissions.

Here we present a new, dynamic method for estimating the soil CO_2 balance of drained peatland forests in Finland, on time scales needed in the GHG inventory. We use both measured and modelled values of peat and litter decomposition and plant litter input. Water table depth (WTD) is the main factor that controls decomposition in drained wetlands (Silvola et al. 1996, Couwenberg et al. 2011, Jauhiainen et al. 2019, Evans et al. 2021), but direct data of WTD is not nationwide available for GHG inventory purposes in Finland. We therefore link decomposition of peat and litter to tree basal area per hectare (BA)

using empirical regression models available for different drained peatland forest site types (Ojanen et al. 2014). BA provides a proxy of the rate of evapotranspiration, which largely controls WTD in forestry-drained peatlands (Hökkä et al. 2021, Leppä et al. 2020). In addition, BA can be used to predict litter input from trees and ground vegetation (Ojanen et al. 2014). The time series of soil CO_2 balance is then calculated using a time series of BA provided by the Finnish National Forest

70 Inventory, NFI (Tomppo et al. 2011, Korhonen et al. 2021), combined with a time series of air temperature. Since drained peatland forests are not evenly distributed across Finland, the temperature dependent peat and litter decomposition is modelled using spatially matched environmental variables. With this method, soil CO_2 balance can be estimated as a difference of C added to soil via litter input and CO_2 lost through SOM decomposition (Ojanen et al. 2012).

The CO_2 balance calculation method that was earlier applied in the Finnish GHG inventory for drained peatland forest soils

75 is also dynamic, but only in terms of belowground litter input. The estimate of SOM decomposition is static and spatially poorly representative, as it relies on data collected from a very limited number of field sites (Minkkinen et al. 2007). Aboveground litter input is not included in the calculation. Developing a new, fully dynamic method became possible when new data and empirical models appeared that were able to link together the drivers and components of soil CO_2 balance (Ojanen et al. 2010, 2012, 2013, 2014). To assess the consequences of adopting the new method into the GHG inventory of

80 Finland, we also present, for comparison, the CO_2 balance of drained peatland forest soils as estimated by the old method.

2 Materials and methods

2.1 The concept of soil CO2 balance

The estimation of soil CO_2 balance is based on the annual difference between the release of CO_2 from decomposing soil organic matter (SOM), or heterotrophic soil respiration R_{Het}, and the C entering the soil through plant litter input, and can be

85 expressed as:

$$CO_{2\,Net} = R_{Het} - \frac{44}{12}\left(I_{AGL} + I_{BGL} + I_{AGR} + I_{BGR}\right) \tag{1}$$

where I_{AGL} and I_{BGL} are the annual C input of aboveground and belowground litter, respectively, from living trees and ground vegetation; I_{AGR} and I_{BGR} are the annual C input of aboveground and belowground residues, respectively, from forest harvests and naturally dying trees, and 44/12 is the ratio of CO_2 and C molar masses, converting the organic C in dead organic matter

90 to the unit of CO_2. Positive results in Eq. 1 appear when decomposition R_{Het} is larger than the input of litter and residues and denote net CO_2 emission from soil to the atmosphere. Negative values denote net removal of CO_2 from the atmosphere. The time dimension of CO_2 balance is one year, and decomposition therefore includes wintertime decay as well.

R_{Het} can further be defined as the sum of peat decomposition and decomposition of aboveground and belowground plant litter and residues from harvests and naturally died trees:

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$$R_{Het} = D_P + D_{AGL} + D_{BGL} + D_{AGR} + D_{BGR} \qquad (2)$$

where D_P is the annual CO_2 release from decomposing peat, D_{AGL} and D_{BGL} are the annual CO_2 release from decomposing aboveground and belowground litter, respectively, of living trees and ground vegetation, and D_{AGR} and D_{BGR} are the annual CO_2 release from decomposing aboveground and belowground harvest and natural mortality residues, respectively.

2.2 Drained peatland forest site types

One key element in our method is that litter input and decomposition are individually estimated for each drained peatland forest site type, hereafter denoted as "FTYPE". In Finland, FTYPE classification was developed for guiding drainage and forest management on peatlands. It follows the general Finnish forest site type theory (Cajander 1913) and uses species of trees and ground vegetation as indicators of moisture and fertility regimes. This typification of drained peatlands (Laine 1989, Laine et al. 2012) has been a practical tool for forestry and forest inventory in Finland. Forest growth and the rate of SOM decomposition are related to FTYPE: more fertile FTYPEs support greater tree biomass and growth, and consequently also have higher decomposition rates (Silvola et al. 1996, Minkkinen et al. 2007, Ojanen et al. 2010).

The current typology of FTYPEs follows the principles presented in Laine et al. (2012) and includes a total of five FTYPEs: i.e., herb rich FTYPE, *Vaccinium myrtillus* FTYPE, *Vaccinium vitis-idaea* FTYPE, dwarf shrub FTYPE and *Cladonia* FTYPE, listed in order of decreasing soil fertility (Table 1). Differences in fertility between FTYPEs are evident in tree and ground vegetation species composition (Table 1). We did not separate the two subtypes of the *V. myrtillus* and *V. vitis-idaea* FTYPEs (I and II with different original mire types; Laine et al. 2012), because field determination of especially *V. vitis-idaea* FTYPE subtypes can be ambiguous. Drained peatland forest sites have been classified into FTYPEs in field surveys in all NFIs, but only since NFI10 (carried out in 2004-2008) the classification, shown in Table 1, has been employed. The FTYPEs used in NFI8 (1986-1994) and NFI9 (1996-2003) were converted to correspond to the classification in Table 1 using field collected data on soil type, drainage status, site fertility and vegetation.

The areas and proportions of FTYPEs of all drained peatland forests remaining forest in southern and northern Finland, derived from NFI12 (2014-2018) according to GHG inventory requirements, are shown in Table 1 and the distribution of FTYPEs across Finland is illustrated in Fig. 1 (the definition of southern and northern Finland used in this paper follows the definition used in NFI). FTYPE areas have changed along the years due to changes in land use, such as new peatland drainage or reclamation of agricultural land. To provide a time series of FTYPE areas for the GHG inventory period starting from 1990 (Fig. 2), area data were collected from six NFIs, i.e., NFI8 (1986-1994), NFI9 (1996-2003), NFI10 (2004-2008), NFI11 (2009-2013), NFI12 (2014-2018) and NFI13 (2019-). As for all other time series that utilize NFI data, the years between NFI mid-years are interpolated. However, the years 1990-1992 in northern Finland are extrapolated because the NFI8 mid-year was after 1990, and from 2017 onwards the values are determined with the help of the not yet complete data

from NFI13. The development of Finnish NFI until NFI9 is described by Tomppo et al. (2011) and the sampling design of NFI12 by Korhonen et al. (2021).

The drained peatland forest sites included in our data are those classified as forests in the Finnish FAO Global Forest Resources Assessment (FRA) (FAO 2018): i.e. the area is ≥0.5 ha, trees are able to reach height of 5 m and the canopy cover is >10 %. Currently, NFI field surveys include FRA land classification, and the earlier datasets from 1990s have been reclassified by NFI.

2.3 Climatic variables

The spatial distribution of FTYPEs (Fig. 1), and thereby their climatic attributes, differ due to those geographic reasons that originally resulted in the formation of different mires in Finland (Ruuhijärvi 1960, Havas 1961, Eurola 1962). Therefore, the FTYPE means of May–October mean air temperature (approximating the growing season mean air temperature), which are needed to predict SOM decomposition in different FTYPEs (see Table 6), are calculated using real FTYPE distributions in southern and northern Finland (Fig. 1). Briefly, the May–October mean air temperature is first produced for each drained peatland forest location by combining the location data of sample plots in NFI12 with a 10 km × 10 km weather grid data provided by the Finnish Meteorological Institute (Venäläinen et al. 2005). Using the obtained data, FTYPE mean values are calculated for southern and northern Finland. For simplicity, NFI12 locations are used to calculate the FTYPE mean temperatures for all years in the 1990-2021 time series (i.e. we assume that FTYPE distributions have not significantly changed in time). Lastly, the time series for both regions of the country are smoothed using 30-yr rolling means (Fig. 3).

Decomposition, i.e. CO_2 production of litter from forest harvests and naturally died trees is calculated using the Yasso07 model (Tuomi et al. 2009, 2011), which uses mean annual temperature, temperature amplitude (the difference between the mean temperature of the warmest and coldest month divided by two), and annual precipitation as environmental drivers. Spatially weighed and smoothed 1990-2021 time series for these variables are produced as for May–October mean air temperature above, but since the harvest residue data is currently allocated between southern and northern Finland, not to each FTYPE, the means are calculated for southern and northern Finland, not for each FTYPE (Fig. 3, Supplementary Fig. 1).

2.4 Basal area

FTYPE mean basal area per hectare, BA (Fig. 2) is used to predict peat and litter CO_2 release (see Table 6) and ground vegetation and arboreal fine root litter input (Tables 4, 5). The FTYPE mean BA is estimated using data of tree stem diameter (measured at breast height, DBH) of tree species groups (pine, spruce and deciduous species), collected in the sample plots of the NFI8–NFI13 inventories. As in the case of FTYPE areas, FTYPE mean BAs are interpolated for years between the NFI mid-years, extrapolated for 1990–1992 in northern Finland and derived from the not yet complete NFI13 for years 2017–2021 to produce the time series needed in GHGI (Fig. 2). BAs have generally increased with time in drained peatlands, demonstrating the benefits of drainage for tree growth (Fig. 2).

2.5 Calculating litter input

Plant litter input consists of (1) litter from living trees (excluding fine root litter), (2) arboreal fine root litter (roots of trees
160   and dwarf shrubs with ≤2 mm diameter), (3) ground vegetation litter (excluding dwarf shrub fine root litter) and (4) litter
originating from forest harvests and natural tree mortality. In all calculations, all litter is assumed to contain 50 % C of dry
mass.

To estimate litter input from living trees, the mean biomass (Mg ha^{-1}) of each tree component (stem wood, stem bark, alive
branches, dead branches, foliage, base/stump, and coarse roots of >10 mm diameter) for each FTYPE is first obtained from
165   NFI. These biomasses are calculated by multiplying mean stem volume in each FTYPE with biomass conversion and
expansion factors (BCEFs), based on biomass models of Repola (2008, 2009). BCEFs are component-biomass to stem-
volume ratios, estimated using a sub-sample of trees measured in detail in each NFI for all different combinations of tree
species group, FTYPE and North/South region. Inputs of litter from living trees, consisting of senesced foliage, branches,
stem and stump bark and coarse roots (>10 mm diameter), are then estimated by multiplying the biomass estimates and litter
170   production rates (Table 2) of tree components. Litter production rate tells the proportion of the component mass that turns
into litter in a year.

To calculate arboreal fine root litter input, fine root biomass is first estimated for each FTYPE by using FTYPE mean BAs of
tree groups and FTYPE mean dwarf shrub cover (Table 3) as predictors in empirical regression models (Table 4), obtained
from Ojanen et al. (2014). Trees and dwarf shrubs are combined due to difficulties in distinguishing their fine roots in field
175   samples. The dwarf shrub cover for the *Vaccinium myrtillus* and *Vaccinium vitis-idaea* FTYPEs are calculated as weighted
means of values in I and II sub-types with the relative areas of sub-types (61.3 % and 38.7 % for the *V. myrtillus* FTYPE and
60.1 % and 39.9 % for the *V. vitis-idaea* FTYPE, respectively) as a weight factor. Arboreal fine root litter input is then
estimated by multiplying fine root standing biomass by tree fine root turnover rate measured for each FTYPE (Table 3).

Turnover rates of tree fine roots in Table 3 were estimated using minirhizotrons installed in each FTYPE (except for the
180   *Cladonia* FTYPE, for which the estimate of the dwarf shrub FTYPE is used). Following the minirhizotron-method (Lukac
2012), transparent acrylic tubes (9–12 tubes per site) were installed into the soil on six sites to photograph fine roots
(Minkkinen et al., unpublished manuscript). After a one-year stabilisation period, all tubes were photographed 19 times
during four consecutive years. Images of 1.3 cm × 2 cm (height × width) were taken with a minirhizotron camera (BTC-2;
Bartz Technology, Santa Barbara, USA) and analyzed using a WinRHIZO Tron 2015b program (Régent, Canada). The
185   diameters, lengths and depths of roots were recorded. Root longevities (1/turnover) were determined as years with the
method by Kaplan and Maier (1958) using median longevity, i.e., the lifetime when 50 % of the observed roots had died. If
50 % mortality was not reached during the monitoring period, a parametric regression model with Weibull error distribution
was applied to predict it using survreg (Kalbfleisch & Prentice 2002) in R-package Survival 3.4-0. The turnover rates are
based on tree fine roots with ≤0.5 mm diameter, and they are applied for all arboreal fine roots of ≤2 mm diameter.

190 The mean combined aboveground and belowground litter input from ground vegetation (excluding dwarf shrub fine root litter input) is calculated for each FTYPE using FTYPE mean BA as a predictor in empirical regression models (Table 5), obtained from Ojanen et al. (2014).

In forest harvests, most of the collected roundwood is utilized for wood products, but some is, together with harvest residues, used as energy wood. Therefore, to calculate litter input from harvest residues, a chain of calculations is needed. (1) Of all

195 harvested roundwood volume in Finland (luke.fi/en/statistics/), the proportion that is collected from drained peatland forests is first derived using the respective share of harvest area data provided by the NFI. (2) This proportion of roundwood volume is then converted to total tree biomass, and (3) using data provided by Luke Statistics (2022) of energy wood consumption, the part of biomass that is used for energy production is deducted from the total harvested biomass. (4) The remaining biomass is finally converted to different tree biomass components using specific BCEFs. Of these components, the foliage,

200 branches, waste wood, stumps and coarse roots are assumed to remain in the harvest sites, and thus to compose the harvest residue litter input.

The yearly production of litter from naturally died trees is derived from NFI. For these trees, litter input consists of all biomass components, including the stem wood, calculated using appropriate BCEFs. Data of litter input from harvested and naturally died trees are currently available for southern and northern Finland, but not for each FTYPE separately.

205 Finally, litter input lacks one biomass component, i.e. tree roots in the diameter range of 2–10 mm. The reason is that there is neither biomass nor turnover rate estimate available for these roots.

2.6 Calculating soil CO$_2$ release, R_{Het}

Annual CO$_2$ release from decomposing peat, D_P, and decomposing aboveground and belowground litter of living trees and ground vegetation, D_{AGL} and D_{BGL}, is calculated using empirical regression models given in Ojanen et al. (2014). These

210 models, presented in Table 6, include a FTYPE-specific constant and FTYPE May–October mean air temperature (Fig. 2) and FTYPE mean BA (Fig. 3) as dynamic predictors. The data behind the regression models include field measurements on 68 drained peatland forest sites (Ojanen et al. 2010, 2013), which cover the distribution of FTYPEs in Finland (see Fig. 1 in Ojanen et al. 2010). The field measurements were originally performed on trenched field plots (with cut plant roots to avoid autotrophic respiration) where the aboveground litter was removed, but the CO$_2$ release from decomposing aboveground

215 litter was later calculated using the Yasso07 decomposition model and added to the recorded values of CO$_2$ release (Ojanen et al. 2013). The predicted CO$_2$ release is a yearly estimate although the temperature predictor is the mean May–October air temperature (Ojanen et al. 2014).

The soil CO$_2$ release estimates produced by the regression models in Table 6 do not include CO$_2$ release from harvest residues (the empirical data for the models was collected at sites without recent harvests) or from stump and stem wood of

220 naturally died trees (because these parts of litter did not fit inside the CO$_2$ flow measurement chambers). The CO$_2$ release from these litter components is therefore estimated using the Yasso07 decomposition model, combined with time series of

southern and northern Finland mean annual air temperature, temperature amplitude and mean annual precipitation (Fig. 3, Supplementary Fig. 2). Since the Yasso07 model produces the remaining OM pool after decomposition, the CO_2 release from decomposing aboveground and belowground harvest residues and stump and stem wood of recently naturally died

225 trees, D_{AGR} and D_{BGR}, is calculated by subtracting the remaining OM pool from the inputs of these residues, I_{AGR} and I_{BGR}.

The Yasso07 model was first initialized for 50 years using constant residue input from harvests and naturally died trees (mean of 1970-1976 input for southern Finland and mean of 1975-1977 input for northern Finland). The initialized model was then run from 1971 onwards using real annual residue input. During the 50-yr initialization, the OM pool originating from harvest residues and naturally died trees was monotonically approaching a plateau (Supplementary Fig. 2). The

230 rationale for using unusually short spin-up is that there is no long history for drained peatland forests in Finland (drainage activity rose after 1940s and was most intensive in 1970s) and the soil OM pool originating from anthropogenic activity has not been developing for long. Running longer spin-up would create unnaturally high harvest and natural tree mortality-based soil OM pool, leading to erroneously high CO_2 release from these, slowly decomposing C stocks during the GHG inventory reporting period. However, to be aware of the effects of the short spin-up, we also initialized the Yasso07 model for 1000

235 years. Increasing the length of initialization from 50 to 1000 years increased CO_2 release along the years 1990–2021 annually by 2.4–0.6 % (mean 1.4 %) and 4.1–1.5 % (mean 2.8 %) in southern and northern Finland, respectively.

2.7 Uncertainty analysis

Uncertainty was assessed for each annual estimate of soil CO_2 balance, as well as for the estimate of change in balance between years 1990 and 2021. The accounted sources of uncertainty included NFI sampling errors in area and BA estimates,

240 estimation error in the parameters of the models, litter production rates and dwarf shrub coverage, and uncertainty about litter production originating from living trees, harvests, and natural mortality. The details of variance propagation are given in the Appendix 1. These and all other computations were carried out in the R environment (R Core Team 2020).

2.8 Old calculation method

The old calculation method of drained organic forest soil CO_2 emissions earlier applied in the GHG inventory of Finland

245 (Statistics Finland 2022) differs in many ways from the new method we present here. In the old method, the estimates of heterotrophic soil respiration in FTYPEs are based on results from two peatlands, located at 61°N 25°E (covering FTYPEs from the herb rich to dwarf shrub FTYPE) and 62°N 31°E (representing the *Cladonia* FTYPE) (Nykänen et al. 1998, Minkkinen et al. 2007). The estimates include D_P and D_{BGL} only, are static in time and do not make a difference between the southern and northern Finland. Belowground litter input from living trees, ground vegetation, harvest residues and natural

250 mortality is included, whereas aboveground litter input (and most of its decomposition) from living plants and harvests is excluded. Also, in contrast to the new method, the belowground litter input from ground vegetation is constant, 4.0 t CO_2 ha⁻

[1] yr^{-1} (Laiho et al. 2003), and the arboreal fine root turnover rate is 0.85 for all FTYPEs (Liski et al. 2006). Also, unlike in the new method, tree leaf biomass is used as a proxy to estimate arboreal fine root biomass (Helmisaari et al. 2007).

3 Results

3.1 Soil CO_2 balance

The method developed in this study produces a generally increasing trend of emissions from 1.4 to 7.9 Mt CO_2 for drained organic forest soils in Finland for the period 1990–2021 (Fig. 4). The difference between years 1990 and 2021 is statistically significant (uncertainty less than 100 %; Table A10 in Appendix 1). Across the period 1990–2021, emissions are on average 3.4 Mt and -0.3 Mt in southern and northern Finland, respectively (Fig. 4). Considering the uncertainty, drained organic forest soils in southern Finland and in the whole country are a source of CO_2 in 2021, while the 2021 net emission in northern Finland does not significantly differ from zero (Table A9 in Appendix 1).

In contrast to the results produced by the new method, the old GHGI method produces a decreasing trend of emissions from 12.1 to 3.2 Mt CO_2 for the period 1990–2021 for the whole country and the emissions are on average higher in northern than southern Finland (Fig. 4).

3.2 Total litter input and decomposition

The new method produces an increasing trend of litter input and decomposition for both southern and northern Finland (Fig. 5). In 2021, total litter input and decomposition are, per unit area, 28 % and 34 % higher in southern than northern Finland, respectively (Fig. 5). In both regions, the increasing trend in total litter input is mainly due to increasing input of aboveground tree litter and belowground arboreal litter (Fig. 5). These litter fractions also include residues from forest harvests and naturally died trees.

When compared to the old method in 2021, the new method produces 61 % and 37 % smaller belowground arboreal litter input, per unit area, for northern and southern Finland, respectively (Fig. 5). However, total litter input is on average 32 % and 62 % higher, respectively, because aboveground tree litter and ground vegetation litter are included in the new method (Fig. 5). In 2021, the new method produces 10 % and 63 % higher CO_2 release from decomposition, per unit area, than the old method for northern and southern Finland, respectively (Fig. 5).

3.3 Input and decomposition of litter from harvested and naturally died trees

With the new method the input of litter from harvested and naturally died trees, as well as the CO_2 release from their decomposition, are significantly higher, per unit area, in southern than northern Finland but increase with time in both regions (Fig. 6). The CO_2 balance of these residues is negative (thus acting as a sink) and has a decreasing overall trend in

both regions (Fig. 6). In 2021, the residues form 20 % and 15 % of all litter input, and their decomposition 15 % and 12 % of total decomposition in southern and northern Finland, respectively (compare Figs. 5 and 6).

3.4 Comparison of FTYPEs

When litter input and decomposition (excluding litter from harvested and naturally died trees) are estimated with the new

285   method for each FTYPE individually, both litter input and decomposition have a general decreasing trend along the FTYPE fertility gradient from the most fertile herb rich FTYPE (Rhtkg) to the least fertile *Cladonia* FTYPE (Jatkg) (Fig. 7). The only exception in this pattern is litter input in the *Vaccinium vitis-idaea* FTYPE (Ptkg), which in 2021 exceeds litter input in the more fertile *Vaccinium myrtillus* FTYPE (Mtkg) by 20 % and 26 % in northern and southern Finland, respectively (Fig. 7).

290   No clear trend of CO_2 balance emerges along the FTYPE fertility gradient (Fig. 7). Instead, net emissions are higher in both ends of the gradient than in the *Vaccinium vitis-idaea* (Ptkg) and the dwarf shrub (Vatkg) FTYPEs (Fig. 7). In these two FTYPEs, ground vegetation litter input and/or arboreal fine root litter input are relatively high in comparison to other FTYPEs (Fig. 7). In general, the share of ground vegetation litter of total litter input is higher in northern than southern Finland and increases towards the nutrient poor end of the FTYPE fertility gradient (Fig. 7).

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3.5 Sensitivity of soil CO_2 balance to the main dynamic drivers

As temperature, T (and other climatic variables in Yasso07 modelling) and BA are the main dynamic drivers of soil CO_2 balance in our method, we examined their relative importance by producing three sets of scenarios, where either (1) climate, (2) BA (and the associated harvest rate), (3) both or (4) neither (the default method) were fixed to their 1990 values (Fig. 8):

300   the first set is for the CO_2 balance of tree and ground vegetation litter input and SOM decomposition (top row of graphs, excluding harvest residues and litter from naturally died trees), the second set is for the CO_2 balance of input and decomposition of litter from harvested and naturally died trees (middle row) and the third set is for the total soil CO_2 balance (bottom row).

The CO_2 balance of tree and ground vegetation litter input and SOM decomposition (Fig. 8 top row) appears to be sensitive

305   to the increasing T as the scenario with constant BA (i.e. only T increases) closely follows the default scenario. The effects of changes in BA are more subtle: increasing BAs until ca. 2010 (Fig. 2) decrease emissions slightly as the scenario with constant climate (only BA changes) produces less CO_2 than the scenario where both BA and climate remain constant. From ca. 2010 onwards BAs in FTYPES level off or start decreasing (Fig. 2) and the BA effect turns neutral (northern Finland) or slightly positive (southern Finland) (Fig. 8 top row).

310   The CO_2 balance of harvest residues and litter from naturally died trees (Fig. 8 middle row) is driven by changes in BA and harvest rate as the scenario with constant climate (only BA and harvest rate change) closely follows the default scenario.

Changes in climate increase emissions slightly as the scenario with constant BA and harvest rate (only climate changes) produces more CO_2 than the scenario where BA, harvest rate and climate all remain constant.

The scenarios for the total soil CO_2 balance (Fig. 8 bottom row) combine the main characteristics of the scenarios for the two
315   parts of the total balance: i.e. that the shape and the positive slope of CO_2 emissions in the default scenario are driven by climate warming, and that the main result of increasing BA and harvest rate from 1990 onwards is a lower level of net soil CO_2 emissions.

3.6 Ecosystem CO_2 balance

320   The ecosystem CO_2 balance in drained peatland forests is governed by the combination of soil CO_2 balance and the CO_2 sink created by living trees. When the soil CO_2 balance results of the new method are combined with tree CO_2 sink for southern Finland, the ecosystem CO_2 balance is negative, i.e. the forests are a CO_2 sink, until the beginning of 2010s when the balance turns positive (Fig. 9). This change of pattern is a product of both lower tree CO_2 sink in 2010s and increasing soil CO_2 emissions (Fig. 9). In northern Finland, drained peatland forests are still a CO_2 sink, but also there the gradually decreasing
325   tree CO_2 sink together with increasing soil CO_2 emissions creates a decreasing trend for the ecosystem sink in 2010s (Fig. 9).

4 Discussion

4.1 The CO_2 balance of drained organic soils by the new method

The CO_2 balance of drained organic soils in Finland, as estimated by the new method we describe in this paper, has two main features (Fig. 4): first, drained peatland soils are currently a source of CO_2 in Finland, and second, the emissions are
330   increasing with time due to the high sensitivity of SOM decomposition to increasing temperature. The increase of 1.2 °C in mean annual temperature across the time series, visible in both southern and northern Finland (Fig. 2), increases CO_2 emissions from soil by 8.1 Mt CO_2 in the whole country (the constant BA and harvest scenario in Fig. 8 bottom row). An increase in BA and harvest rate over the decades counteract this pattern (the constant climate scenario in Fig. 8 bottom row), however, and the increase in net soil CO_2 emissions across the time series remains smaller (the default scenario in Fig. 8
335   bottom row).

Both litter input and decomposition have increasing trends in time (Fig. 5). The increasing trend in litter input is mainly driven by the general increase of BA across the FTYPEs (Fig. 3), which leads to greater litter input from living trees (evident in both aboveground tree litter input and belowground arboreal litter input in Fig. 5). Part of the increasing trend is also explained by the increasing input of litter from harvested and naturally died trees (Fig. 6). Litter input from ground
340   vegetation, instead, remains rather stable across the time series (Fig. 5). The increasing CO_2 release from decomposition is driven by both increasing BA and increasing May-October temperature (Fig. 2) as both are positively associated with decomposition in the empirical data (Table 6; Ojanen et al. 2014). Part of the increasing trend is also explained by the increasing CO_2 release from the decomposing litter of harvested and naturally died trees (Fig. 6). The likely reasons why BA

is positively linked to decomposition in the regression models by Ojanen et al. 2014 (Table 6) is that increasing litter input

345   sustains larger decomposer biomass and thus higher CO_2 production, and that higher BA - through higher evapotranspiration – maintains lower soil water table level and a deeper oxic layer for SOM decomposition (Sarkkola et al. 2010). When the impacts on litter input and decomposition are combined, increasing BA first slightly decreases net soil CO_2 emissions (scenario of constant climate in Fig. 8 top row), but this effect disappears in mid 2010s, when the positive trend in BA levels off or turns negative in most FTYPEs (Fig. 2).

350   Release of CO_2 from decomposition per area of drained peatland forest soil is significantly higher in southern than northern Finland (Fig. 5). This is due to higher temperature, but also due the generally higher BA and litter production (Fig. 2, 3, 5) in southern Finland. Input of litter from forest harvests has an increasing trend, and in the timescale of the inventory time series, harvest residues create a CO_2 sink and decrease net soil CO_2 emissions (Fig. 6). However, this does not imply that net ecosystem CO_2 emissions would decrease with harvests because tree CO_2 sink simultaneously decreases (see trends in 2010s

355   in Fig. 9).

Of the FTYPEs, the dwarf shrub (Vatkg) and *V. vitis-idaea* (Ptkg) FTYPEs are clustered to the water divides (Suomenselkä and Maanselkä) of the southern and northern regions of the country, while other FTYPEs are more evenly distributed across the country (Fig. 1). These two common FTYPEs have relatively low nutrient status (Table 1) and both have, even without the harvest residue impact, for most part of the GHG inventory time series a negative or near zero soil CO_2 balance in the

360   North, and the *V. vitis-idaea* FTYPE also in the South (Fig. 7). This is roughly in line with earlier findings (Minkkinen et al. 1999, Ojanen et al. 2014). However, the ubiquitous upwards temporal trend of soil CO_2 balance in all FTYPEs (Fig. 7) suggests that also these FTYPEs are entering the phase of net soil CO_2 emissions.

Uncertainty due to model parameters and input data were propagated into the soil CO_2 balance estimates. The parameter estimates of the peat and litter decomposition model were clearly the greatest source of uncertainty, followed by those in the

365   fine-root litter input model (including the turnover rates) and litter input from living trees. Meanwhile, uncertainty in NFI estimates of site type areas and BA had negligible impact on annual balance estimates and somewhat greater, but still minor, impact on the estimates of changes in emissions. This indicates that the proposed method might also be applicable in somewhat smaller regions (with less precise NFI estimates) without drastic increase in uncertainty. Errors in annual soil CO_2 balance estimates are strongly correlated over the whole time series because the same model parameters with the same

370   estimation errors are applied throughout the series. Since all applied models were linear in both their parameters and inputs, it was possible to handle these correlations by combining quantities including the same parameter (Appendix 1). Uncertainty propagation by means of variances relies on an assumption of symmetric error distributions. This assumption is realistic since uncertainty due to all individual components in the balance estimates was 30 % or less, i.e., coefficient of variation was at most 0.15 (Table A9; cf. Frey et al. 2006, sec. 3.2.2.4). Structural uncertainty about the models could not be accounted for

375   since large-scale validation data does not exist.

4.2 Ecosystem CO_2 balance in drained peatland forests

Peatlands in Finland were mostly drained for enhancing tree growth and thus enabling forestry on peatlands. The increasing BAs in drained peatlands from 1990 onwards (Fig. 3) show that this target has largely been achieved. Therefore, when considering the net climate effects of peatland drainage, the development of tree CO_2 sink needs to be taken into consideration (Fig. 8). In 2010s, the trend of gradually increasing BA in drained peatland forests levelled off (Fig. 2). This is due to increasing harvests in these forests (manifested by the increasing harvest residue input in Fig. 6), which in turn is associated with the sharp turn in tree CO_2 sink and net ecosystem balance after 2010, ultimately leading to a shift from a net ecosystem sink to a source of CO_2 in southern drained peatland forests (Fig. 8). After 2010, drained peatland forests have been a sink of CO_2 only in northern Finland, but when the northern tree stands mature for harvesting, and industry demand for wood remains high, the net ecosystem CO_2 sink will likely be lost in northern Finland as well. Calculated for the whole country, forests growing on drained peatlands were a net sink of 0.2 Mt CO_2 in 2021.

4.3 Comparing the soil CO_2 balance predictions by the new and old method

The differences in the estimates of soil CO_2 balance between the new and old GHG inventory methods are remarkable (Fig. 4). The main difference are the opposite trends in time series: in the beginning of the GHG inventory reporting period the new method produces 10.7 Mt lower soil CO_2 balance than the old method, but then reaches 3.4 Mt higher balance in recent years. The mean level of emissions differs a lot in the North, less in the South (Fig. 4).

The contrasting trends of time series produced by the two methods call for explanations. In the old method, CO_2 release from decomposition is constant, whereas in the new method CO_2 release from decomposition increases in time as it is driven by increasing temperatures and BA (Fig. 5). Both methods predict increasing litter input in time due to increasing BA (Fig. 5). In the old method, however, increasing litter input combined with constant decomposition unavoidably leads to decreasing net emissions as the release of CO_2 in decomposition does not follow the increase in litter input. In the new method, both constituents of the CO_2 balance are dynamic, and the method can produce either an increasing or decreasing trend of net emissions depending on the relative magnitude of slopes in trends for litter input and decomposition. The new method predicts that decomposition increases more in time than litter input does, thus leading to an increasing trend in net soil CO_2 emissions.

Besides the difference in the estimation of decomposition, estimation of litter input has major differences between the methods. First, while the above-ground litter (including litter from living trees and ground vegetation, as well as from harvested and naturally died trees) is explicitly included in both litter input and decomposition calculations in the new method, the old method assumes that above-ground litter input and decomposition are in equilibrium, and consequently, above-ground litter is not included in the calculation. However, findings from drained organic forest soils suggest that while increasing litter input is generally associated with increasing decomposition, litter input and decomposition are seldom in equilibrium. For instance, in Finland, litter production exceeds decomposition in the nutrient-poor FTYPEs and vice versa in

the nutrient-rich FTYPEs (Fig. 1 in Ojanen et al. 2013). Other studies also stress the importance of litter input in the C balance of drained peatland forests, and despite plant-community-level litter decomposition rates being somewhat higher in

410   drained than undrained peatland forests, increased tree litter input can still lead to a significant accumulation of new SOM, especially in conifer-dominated stands (Vávřová et al. 2009, Straková et al. 2012). A model-based assessment of forests growing on mineral soils in Finland further suggests that the proportion of C in litter input that remains in the soil C pool is ca. 4 % (Liski et a. 2006), and although these models are developed for mineral soil forests, the main principles of SOM decomposition also likely apply in drained organic soil forests. Finally, as the maturing drained peatland forests are

415   increasingly harvested, litter production from living trees will inevitably decrease and that from harvest residues increase, making it necessary to explicitly include the input and decomposition of above-ground litter in the calculation method.

Another major change in the estimation of litter input in the new method are the FTYPE-specific turnover rates of arboreal fine roots, which constitute the major source of belowground litter. In the light of recent findings, the assumption of a single turnover rate of 0.85, used for all FTYPEs in the old inventory method (Statistics Finland 2021), seems incorrect. The new

420   minirhizotron results included in the new method suggest that tree fine root turnover rate varies markedly by FTYPE and that the turnover rate is on average lower in the nutrient-poor than in the nutrient-rich FTYPEs (Table 4). The lower turnover rate largely explains the remarkably lower arboreal fine root litter production in the new method (Fig. 5).

4.4 Comparing the predicted soil CO_2 balance with other emission factors

The default IPCC emission factors for boreal drained organic soils range from 0.25 (95 % CI -0.23–0.73) t CO_2-C ha^{-1} yr^{-1} in

425   drained nutrient poor forest land to 0.93 (0.54–1.3) t CO_2-C ha^{-1} yr^{-1} in nutrient rich forest land (IPCC 2014; Table 2.1 in Chapter 2). Applying the IPCC default emission factors for the areas of nutrient-poor and nutrient-rich FTYPEs found in southern and northern Finland (Table 1) gives emission factors of 0.56 and 0.49 t CO_2-C ha^{-1} yr^{-1} for these regions, respectively, and an overall factor of 0.53 t CO_2-C ha^{-1} yr^{-1} for the whole country. The emission rate of 0.50 t CO_2-C ha^{-1} yr^{-1} predicted by our method for Finland for the year 2021 is slightly lower, but as the emissions with the new method have a

430   rising trend (Fig. 4), they will likely soon exceed the default emissions.

To compare our results to emission factors used for drained organic forest soils by other European countries, we collected information from the national reporting CRF tables published in 2021. Russia, Norway, Ireland and Poland use emission factors of 0.52–0.72 t CO_2-C ha^{-1} yr^{-1} that are at the lower end of the 95 % CI of IPCC default method emissions for boreal nutrient rich forest land. In temperate areas the IPCC default emission is 2.6 (2.0–3.3) t CO_2-C ha^{-1} yr^{-1} and temperate

435   countries Germany, Denmark, Switzerland and UK apply emission factors of 2–2.6 t CO_2-C ha^{-1} yr^{-1}. As expected, the emissions predicted by our method for Finland in the northern boreal forest zone are lower than the emission factors used in countries that partly or completely locate in the temperate zone.

4.5 Critical assumptions in the new method

It is well established that water table depth (WTD) is the master variable controlling decomposition in drained wetlands
440 (Silvola et al. 1996, Couwenberg et al. 2011, Jauhiainen et al. 2019, Evans et al. 2021), and one can argue that WTD should
be included in methods that aim at predicting CO_2 release from drained peatland soils. In fact, Ojanen et al. (2010), using the
data behind our new method, found that regression models between soil respiration and climate variables had higher
coefficient of determination, R^2 when WTD was included (see Ojanen et al. 2010, Table 5). However, direct data of WTD is
not nationwide available for GHGI purposes in Finland and WTD cannot currently be used as a predictor of decomposition.
445 In undrained peatlands, soil CO_2 balance fluctuates following annual variation in water saturation and temperature (e.g. Alm
et al. 1997, 1999), tree growth is suppressed by anoxia and BA remains low. Draining initiates tree growth and reduces
critical high water level periods, and finally, the increasing tree growth and BA lead to lowering water table level (Sarkkola
et al. 2010). The impact of WTD on decomposition is therefore implicitly included in our method through the impact of BA,
which provides a proxy of the rate of evapotranspiration that largely controls WTD in forestry-drained peatlands (Hökkä et
450 al. 2021, Leppä et al. 2020).

The *in situ* minirhizotron data behind the arboreal fine root turnover rates that we apply are rare, and the results we use are
the first to compare arboreal fine root growth and mortality in different boreal FTYPEs. The measurements carried out with
minirhizotrons focused on tree fine roots with diameter ≤0.5 mm, but we extrapolated the results for all arboreal roots ≤2
mm of diameter. Because thicker fine roots have lower turnover rates, our method may overestimate litter input of roots with
455 a diameter of 0.5–2 mm. In terms of soil CO_2 balance, this would lead to underestimation of net CO_2 emissions. On the other
hand, as the fine root turnover rates of dwarf shrubs are higher than those of trees, at least when compared to Scots pine
(Minkkinen et al., unpublished data), our method may in this part underestimate the arboreal root litter input and
overestimate the net CO_2 emissions.

What may impair the use of BA when estimating SOM decomposition is that relatively little data is available of CO_2 release
460 from peatland forest soils after clear-cut and regeneration, or after other forest management options such as cuttings applied
in continuous-cover silviculture. Mäkiranta et al. (2010) showed that CO_2 release from soil decreased after forest clear-cut
because of rising water table and, on the other hand, because of the dryness of peat soil surface. Instead, CO_2 release from
decomposing harvest residues was high, and because of the removed tree stand and suffering ground vegetation, the site
acted as a high source of CO_2 emissions during the whole study of four years. Similarly, eddy covariance observations in a
465 drained peatland forest showed markedly elevated CO_2 emissions at least two years after clear-cutting (Korkeakoski et al.
2019), thus suggesting significant transitory effects of clear-cutting on soil CO_2 balance in peatland forests. However,
similarly to Mäkiranta et al. (2010) study, a marked share of increased emissions likely resulted from decomposition of fresh
harvest residues also in the Korkeakoski et al. (2019) study, and this harvest residue effect is included in our method in both
litter input and decomposition (Fig. 6).

470   One fundamental assumption in our new method is that the statistical relationships between BA, ground vegetation litter input and SOM decomposition described by Ojanen et al. (2014) for different FTYPEs can adequately predict these associations in forests also under their future successional change. If future forest stands have features that were not present in the studied field sites, this assumption is not valid. This also has relevance in terms of the uncertainty analysis. A strong structural assumption in our new method is that model parameters, including the litter production and turnover rates, do not

475   change in time. If trend-like changes do exist, the reported uncertainties of change estimates are too small (see Lehtonen and Heikkinen 2016 for an illustration). Better understanding of temporal dynamics would be needed to make uncertainty assessments more comprehensive.

Our method also assumes that the spatial covariation in temperature and decomposition, found by Ojanen et al. (2014) among their field sites, is a causal relationship that can be used to describe the effects of increasing temperature on SOM

480   decomposition under climate warming. In our time series, the May–October mean temperature rises by 0.67 °C and the mean annual temperature by 1.2 °C during the GHG reporting period (Fig. 2). This upward trend in temperature increases decomposition in our method (Table 6) and will reduce the C sink of forest ecosystems in drained peatlands if it is not simultaneously accompanied by increasing nutrient mineralization, plant growth and litter input. It is well established that heterotrophic soil respiration is sensitive to temperature (Silvola et al. 1996, Kätterer et al. 1998, Meyer et al. 2018), and

485   while the response to temperature may somewhat attenuate with time due to a gradual decrease of easily degradable C and thermal adaptation of decomposer communities and their respiration (Davidson and Janssens 2006, Bradford et al. 2008, Bradford 2013), it is likely that the trend of increasing SOM decomposition produced by our method represents a real outcome of climate warming. Globally the same impact of increasing temperature on soil CO_2 emissions in drained peatlands is demonstrated by the steeply increasing emission factors from high latitudes to tropics (IPCC 2014; Table 2.1 in Chapter

490   2).

4.6 Further development needs

The impact of WTD on SOM decomposition is implicitly included in our method through BA and T, which both are proxies of the rate of evapotranspiration. However, the relationship between BA and WTD is not linear but attenuating at high BAs (Sarkkola et al. 2010). Furthermore, while ditch spacing and ditch depth have a minor role in comparison to BA, they also

495   affect WTD (Sarkkola et al. 2010, Hökkä et al. 2021), and including models that could link ditch parameters to soil CO_2 release would likely improve our method. Resolving these issues may require transforming the method into a process-based model, which would then also open avenues for addressing a wider range of environmental effects on decomposition.

Also, while our new method covers a wide range of litter input, litter input from tree roots thinner than coarse roots, but thicker than fine roots, i.e. in the diameter range of 2–10 mm is lacking. This is because there are no data of the standing

500   biomass or litter production rate for this fraction of tree roots. In a Swedish study, Petersson and Ståhl (2006) showed that the belowground tree biomass was on average 11 % higher when roots down to 2 mm diameter were included in comparison

to using Marklund's (1987, 1988) biomass equations, which produce the biomass of stump and thicker coarse roots only. Using this estimate, we could assume that we miss ca. 10 % of the tree belowground biomass (i.e. of the combined stump and coarse root biomass) and correct the tree belowground biomass estimate accordingly. However, the turnover rate of this biomass fraction is unknown.

We use 30-year rolling means of climatic variables in our models, computed to the last year of the sequence. This procedure is aimed at smoothing the effects of annual weather variation on the results and thus better reveal the mean level of soil CO_2 balance. However, when the smoothed variable has a monotonous trend in time, which is the case, for instance, for May-October and annual mean temperature that increase in time (Fig. 3), the 30-year rolling mean temperature lags behind the true mean temperature. In our case, this means that the May-October mean air temperature, which largely determines the peat and litter decomposition (Fig. 8 top row), is constantly too low, which in turn leads to too low CO_2 emission estimates. The origin of applying a rolling mean is in the similar solution used in mineral soils. For a more agile smoothed value, another type of filter would be needed.

Finally, while peatlands also release organic C as dissolved and particulate organic carbon, DOC and POC into surrounding waters in their natural stage, drainage increases the release (Nieminen et al. 2021). The IPCC 2013 Wetland Supplement (IPCC 2014) provides default rates of waterborne C release in the form of increased DOC leaching over the natural baseline (IPCC 2014, Ch. 2; Equation 2.5). A simple application of IPCC tier 1 value of 0.12 (0.07–0.19) t C ha^{-1} (IPCC 2014, Ch. 2; Table 2.2), multiplied by the total drained forest land area of 4.3 Mha on organic soil in Finland, gives a rough estimate of peat C loss as DOC of 0.52 Mt C. Using IPCC default assumption that 90 % of DOC is finally released as CO_2 to the atmosphere gives an estimate of 1.70 Mt CO_2 above the soil CO_2 emissions predicted by our new method. This corresponds to a shift of about 21 % of the net soil CO_2 balance in 2021 (Fig. 4), the amount worthy of considering as part of the GHG inventory of drained peatlands. Although the fate of DOC in water courses is not entirely clear (a significant part of DOC likely ends up to fuel CO_2 emissions, but the share of humus that re-sediments to rivers, lakes and sea is difficult to quantify), from the point of view of peatland forest soils, the runaway organic C reduces the peat C pool and displays one more aspect of the anthropogenic impacts of drainage worth of acknowledging.

4.7 Conclusions

Clear merits of our method are that it includes comprehensive empirical data and models for SOM decomposition and litter production and links these to successional changes of tree stand characteristics and to a regionally precise temperature regime. The method also includes realistic propagation of uncertainties in all model inputs and parameters into the annual estimates of CO_2 balance and their differences, which is a requirement for GHG inventories. Such a dynamic model that can consider the effects of both climate change and forest management and development provides a greatly improved tool for forest policy guidance to mitigate climate change.

Conflict of interest

The authors declare that they have no conflict of interest.

Author contribution

Jukka Alm, Antti Wall, Jukka-Pekka Myllykangas, Paavo Ojanen, Juha Heikkinen, Helena M. Henttonen, Raija Laiho, Kari Minkkinen, Tarja Tuomainen and Juha Mikola have jointly developed the concept of the method described here. J-P Myllykangas is responsible for the development of computations, and all authors have participated in writing and cross-commenting the manuscript.

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Figure captions
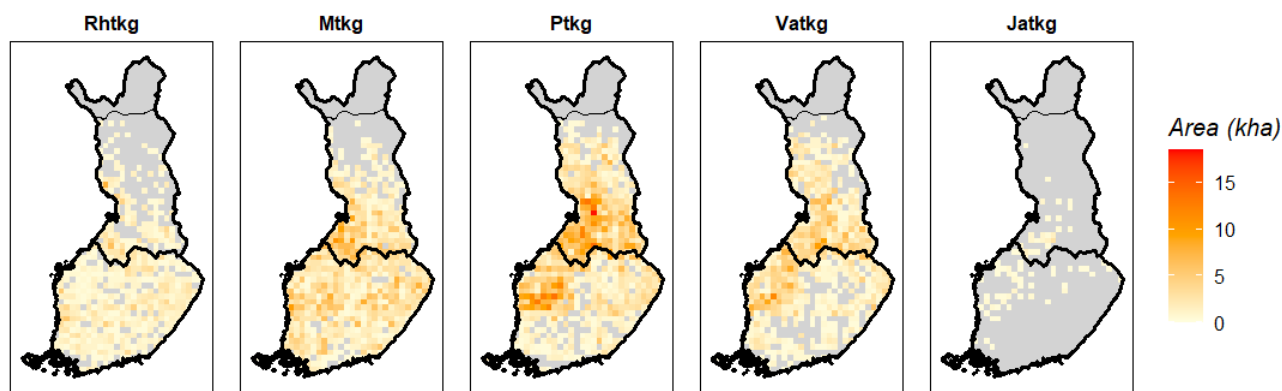
Figure 1. Distribution of drained peatland forest site types (FTYPEs; see Table 1 for abbreviations) in northern and southern Finland, as observed in NFI12. The colour represents the area (kha) of FTYPE in a 20 km × 20 km grid cell and the division of southern and northern Finland (thick black line) follows the division used in the NFI. The northernmost parts of the country (separated with a thin black line) are not included in NFI.

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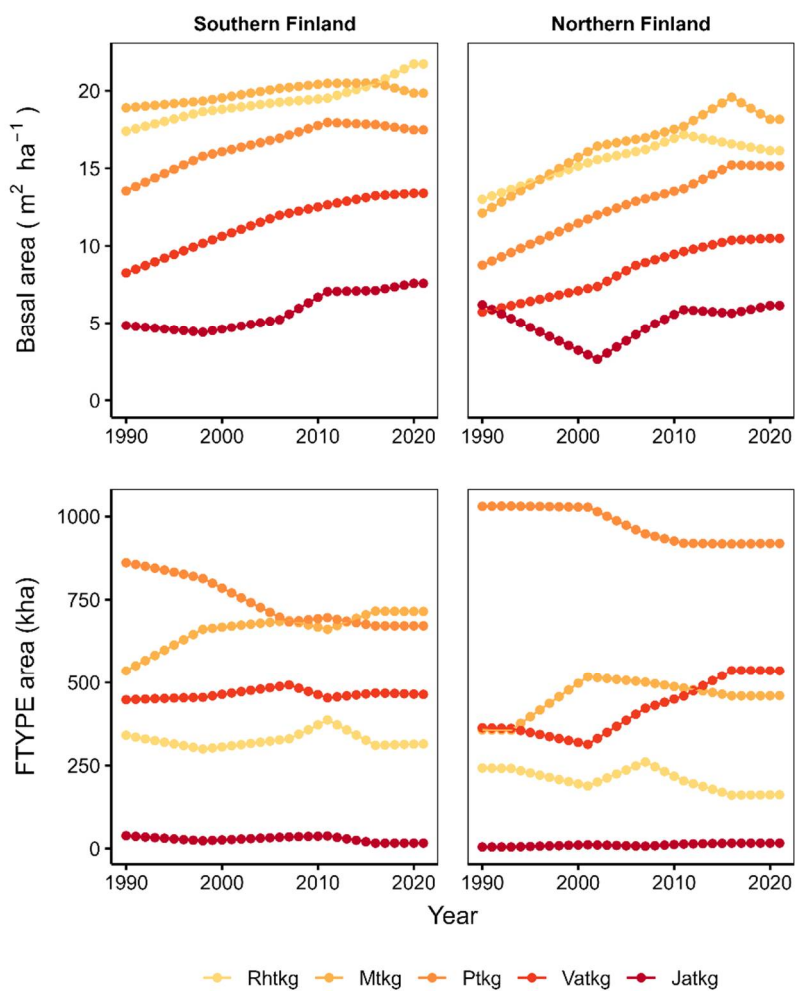
Figure 2. Time series of basal area per hectare for each FTYPE, and FTYPE areas in southern and northern Finland as estimated from NFI data (see Table 1 for FTYPE abbreviations).
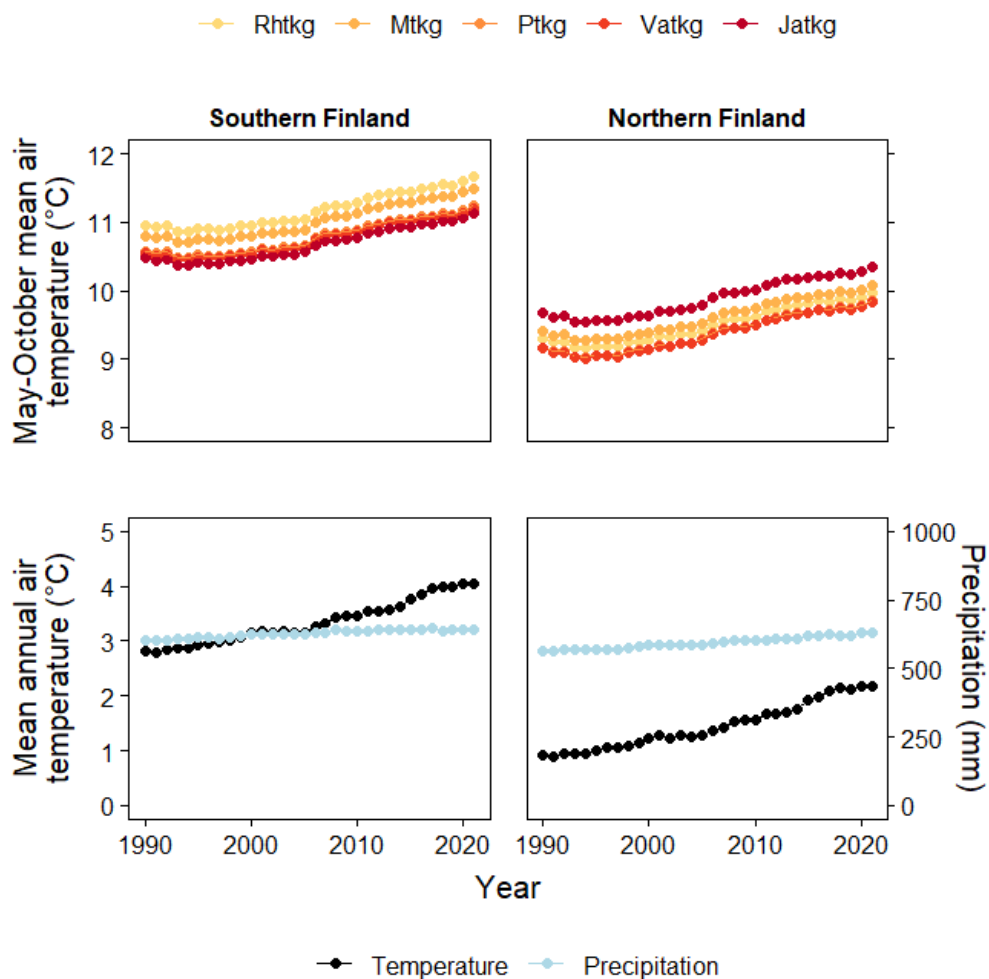
Figure 3. May–October mean air temperature for each FTYPE in southern and northern Finland and mean annual air temperature and precipitation of all drained peatland forests in southern and northern Finland for 1990–2021. In all cases, the means are 30-yr rolling means.
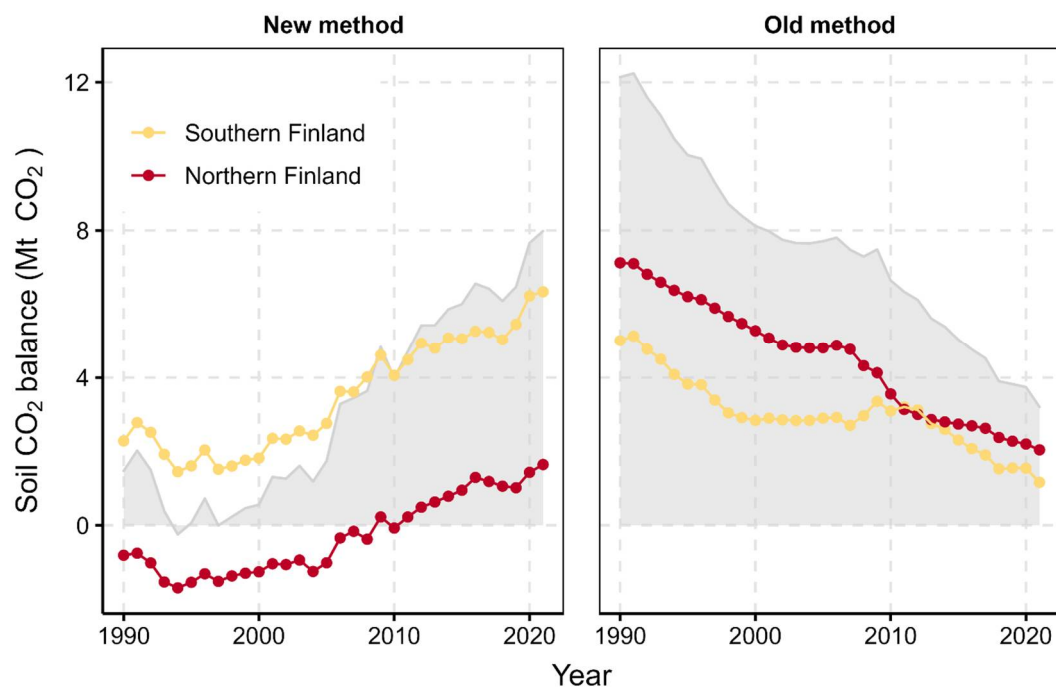
Figure 4. Drained organic forest soil CO₂ balance for the whole country (grey area) and northern Finland (red line) and southern Finland (yellow line) as produced by the new and old method. See Fig. 1 for the two regions of the country.

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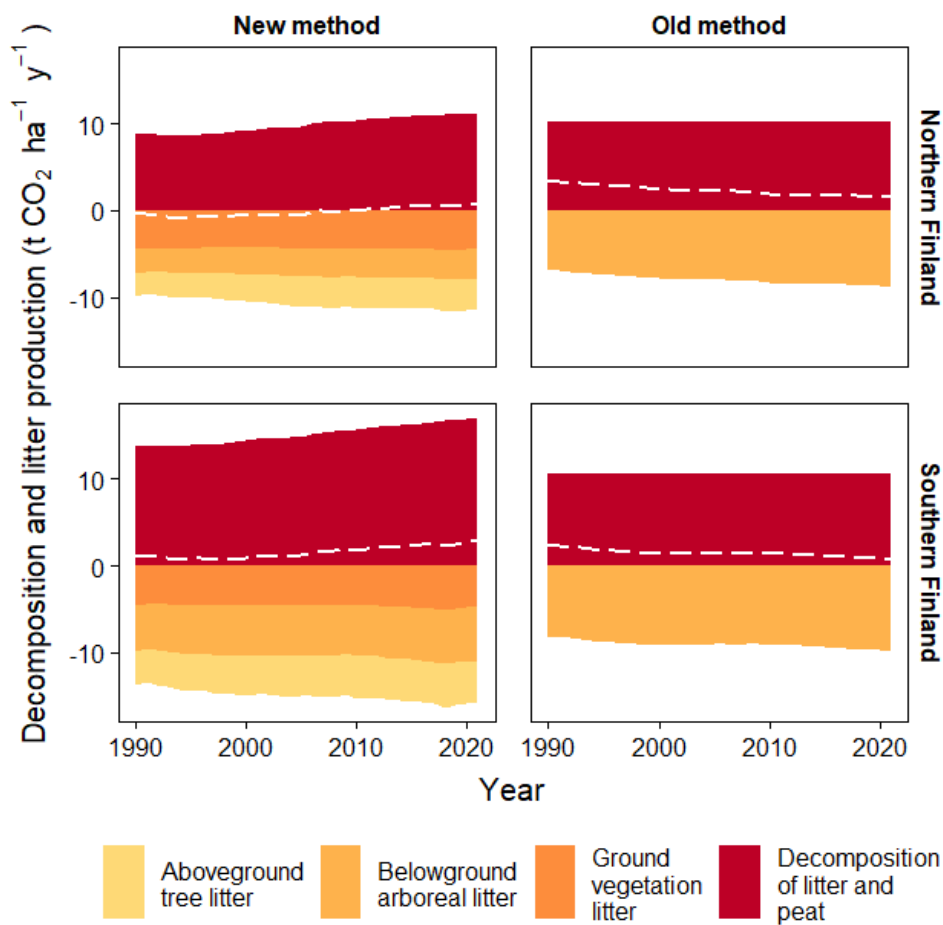
Figure 5. Input of litter fractions, CO₂ release from decomposing litter and peat, and the CO₂ balance (dashed white line) for northern and southern Finland using the new and old method. In the new method, ground vegetation litter consists of both aboveground and belowground litter except for dwarf shrub fine root litter, which is included in belowground arboreal litter.
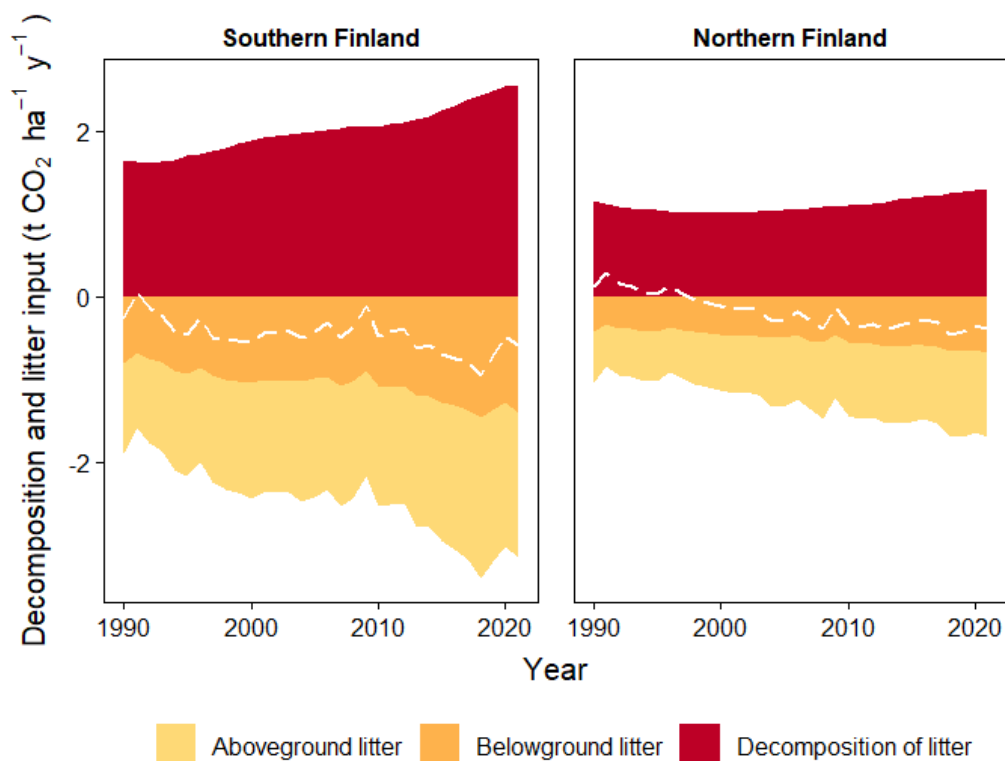
Figure 6. Input and decomposition of litter from tree harvests and naturally died trees, and their CO₂ balance (dashed line) for northern and southern Finland as produced by the new method. Only stem and stump wood are included for naturally died trees.
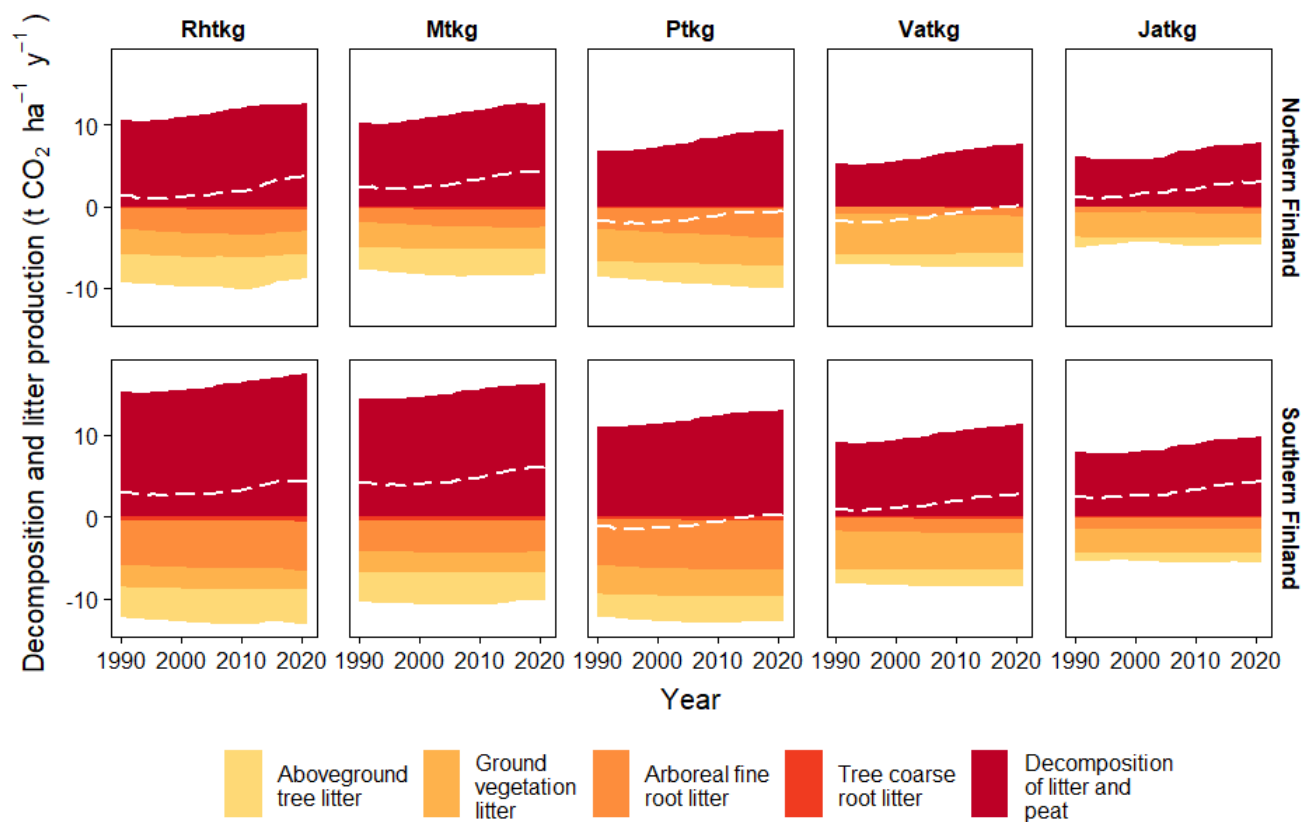
Figure 7. Litter input and decomposition (both exclude input and decomposition of litter from harvested and naturally died trees), and the soil CO_2 balance (dashed line) in FTYPEs in northern and southern Finland as produced by the new method. Ground vegetation litter includes aboveground and belowground litter except for dwarf shrub fine root litter, which is included in arboreal fine root litter. See Table 1 for FTYPE definitions.

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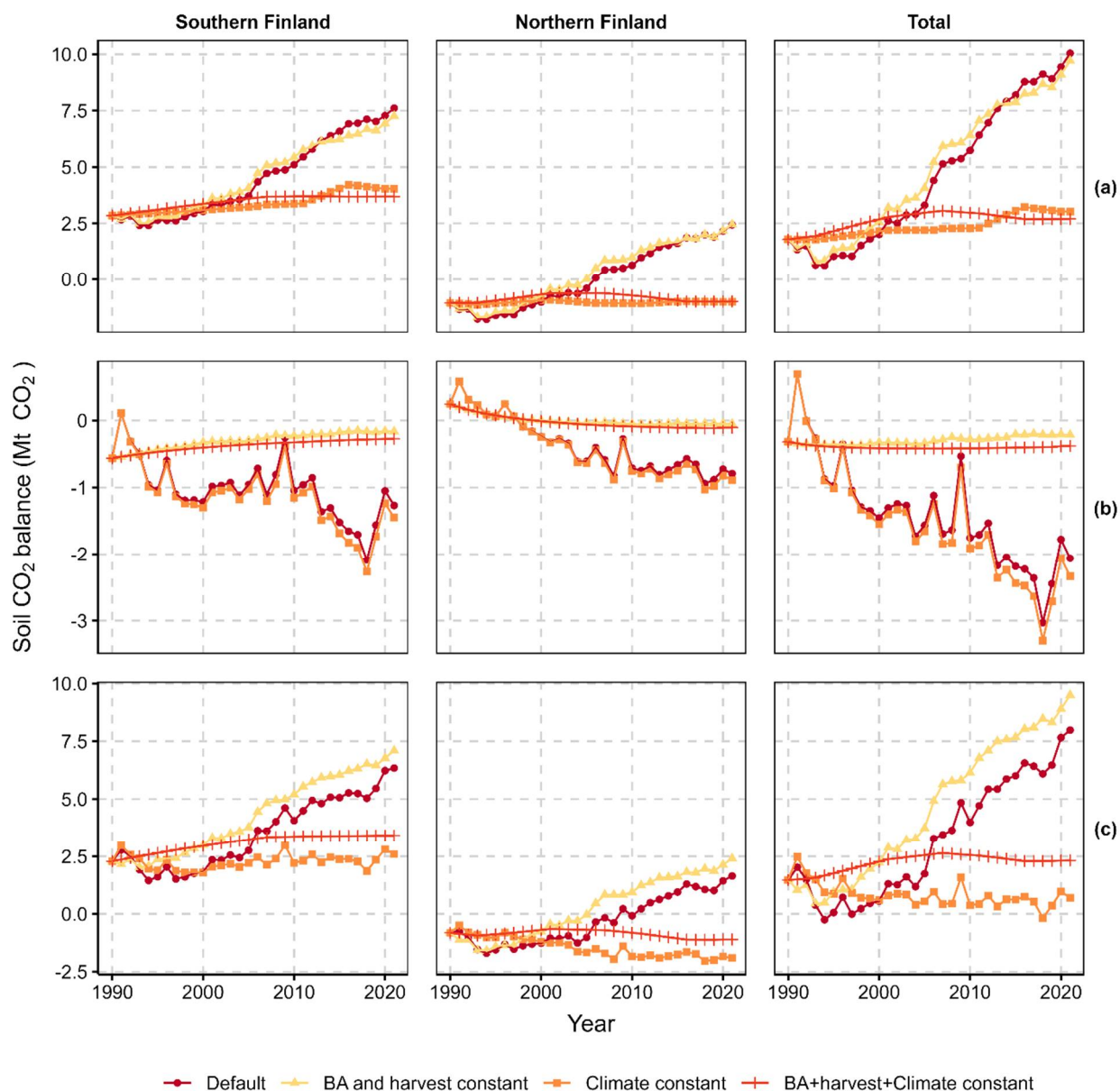
Figure 8. Soil CO₂ balances calculated with four different scenarios for southern and northern Finland and the whole country: "Default" allows all drivers change according to data, "BA and harvest constant" fixes the values of BA and harvest rate to that in 1990, "Climate constant" fixes the growing season mean temperature used by equations of SOM decomposition and the climatic variables used by Yasso07 to values recorded in 1990, and the "BA + harvest + climate constant" fixes all these variables to their 1990 levels. The top row of panels (A) shows the CO₂ balance of live tree and ground vegetation litter input and SOM decomposition (thus excluding harvest residues and litter from naturally died trees), the middle row (B) shows the CO₂ balance of input and decomposition of litter from harvested and naturally died trees, and the bottom row (C) shows the total soil CO₂ balance.

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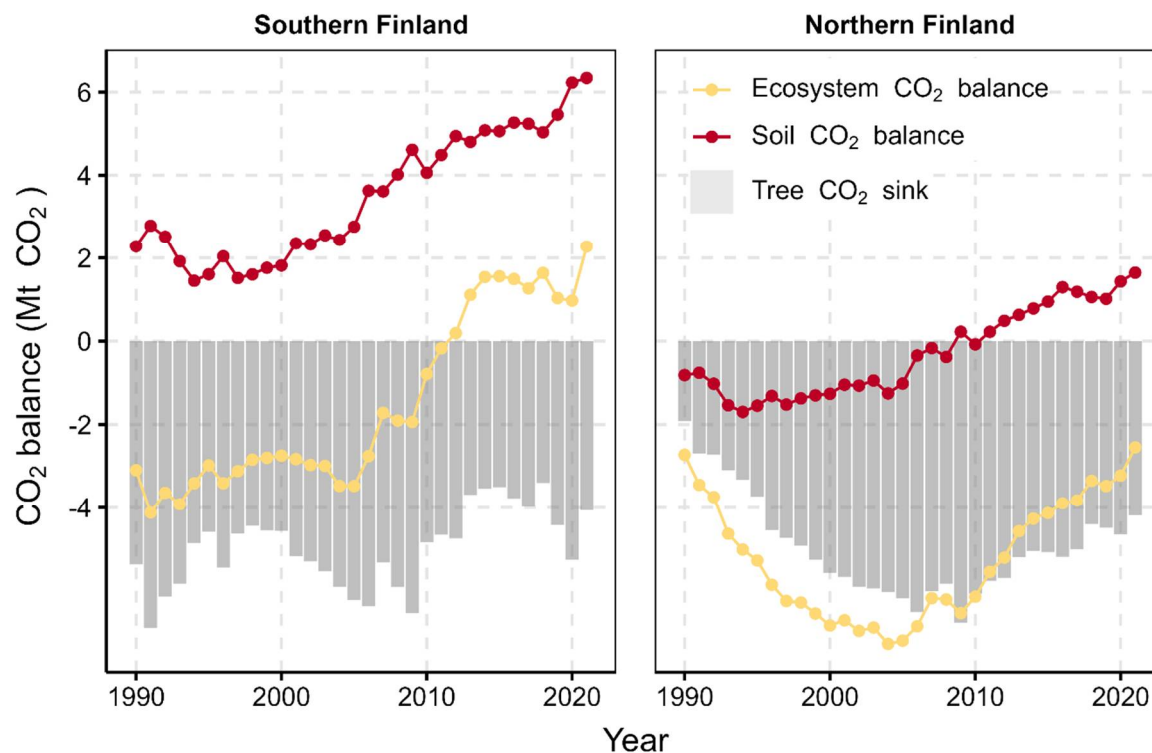
Figure 9. Changes in tree CO$_2$ sink (grey bars), soil CO$_2$ balance (red line) and whole ecosystem CO$_2$ balance (yellow line) (Mt CO$_2$ yr^{-1}) in drained peatland forests in southern and northern Finland across the GHG inventory reporting period (see Fig. 1 for the two regions). Note that no other GHGs than CO$_2$ is included in the calculated balances.

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Tables

Table 1. Characteristics of drained peatland forest site types (FTYPEs) and the GHG inventory estimates of their area and proportion of all drained peatland forest area remaining forest, based on NFI12 data and following the FAO forest classification, in southern and northern Finland (n = number of sample plot centres in each FTYPE). See Fig. 1 for the southern and northern regions of Finland and the distribution of FTYPEs across the regions.

FTYPE	Abbreviation	Dominant tree species	Composition of ground vegetation	Southern Finland	Northern Finland
Herb rich drained peatland forest	Rhtkg	*Betula pubescens, Picea abies*	Herbs with high nutrient requirement such as large pteridophytes, *Oxalis acetocella, Geranium sylvaticum, Cornus suecica, Viola palustris*	0.315 Mha 14 % n = 929	0.162 Mha 7 % n = 319
Vaccinium myrtillus drained peatland forest	Mtkg	*P. abies, Pinus sylvestris*	*Vaccinium myrtillus, Trientalis europaea, Dryopteris carthusiana*	0.715 Mha 33 % n = 2056	0.460 Mha 22 % n = 949
Vaccinium vitis-idaea drained peatland forest	Ptkg	*P. sylvestris*	*V. vitis-idaea, V. myrtillus, Pleurozium schreberi, Dicranum polysetum, Polytrichum commune*	0.672 Mha 31 % n = 1896	0.919 Mha 44 % n = 1811
Dwarf shrub drained peatland forest	Vatkg	*P. sylvestris*	*Rhododendron tomentosum, V. uliginosum, Empetrum nigrum, P. schreberi, D. polysetum*	0.465 Mha 21 % n = 1314	0.536 Mha 26 % n = 1052
Cladonia drained peatland forest	Jatkg	*P. sylvestris*	Large *Cladonia spp.* patches, *Eriophorum vaginatum, Calluna vulgaris, Sphagnum fuscum,* dwarf shrubs	0.017 Mha 1 % n = 47	0.017 Mha 1 % n = 40
			Total	**2.182 Mha n = 6242**	**2.093 Mha n = 4171**

750 **Table 2. Litter production rates of tree biomass components as derived from studies by Lehtonen et al. (2004), Muukkonen and Lehtonen (2004), Starr et al. (2005), Liski et al. (2006) and Ojanen et al. (2014).**

Tree	Foliage	Dead and alive branches	Stem bark	Stump bark	Coarse roots (ø > 10 mm)
Pine	0.33	0.02	0.0052	0.0029	0.0184
Spruce	0.1^S, 0.05^N	0.0125	0.0027	0.0015*	0.0125
Deciduous	0.79	0.0135	0.0029	0.0001	0.0135

[S] Southern Finland, [N] Northern Finland, * Calculated as a mean of pine and deciduous tree estimates as no estimate for spruce was available from literature.

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Table 3. Dwarf shrub cover, obtained from Ojanen et al. (2014), and turnover rate of tree fine roots (≤ 0.5 mm diameter) in FTYPEs.

Drained peatland forest site type	Dwarf shrub cover (% of area)	Tree fine root turnover rate (1 yr^{-1})
Herb rich drained peatland forest	7	0.8
Vaccinium myrtillus drained peatland forest	15	0.5
Vaccinium vitis-idaea drained peatland forest	32	0.7
Dwarf shrub drained peatland forest	45	0.2
Cladonia drained peatland forest	40	0.2

760

Table 4. Regression models from Ojanen et al. (2014) for estimating mean arboreal fine root (≤ 2 mm diameter) biomass (g m^{-2}) for
765 **FTYPEs in southern and northern Finland. BA = mean basal area (m^2 ha^{-1}); decid = deciduous trees; cover$_{shrub}$ = mean dwarf shrub cover (% of area), taken from Table 3.**

Region	Arboreal fine root biomass (g m^{-2})
Northern Finland	$-53.2 + 8.80 \times BA_{pine} + 6.61 \times BA_{spruce} + 17.3 \times BA_{decid} + 4.81 \times cover_{shrub}$
Southern Finland	$120 + 8.80 \times BA_{pine} + 6.61 \times BA_{spruce} + 17.3 \times BA_{decid} + 4.81 \times cover_{shrub}$

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Table 5. Regression models of litter input from ground vegetation (combined dry mass of aboveground and belowground litter excluding dwarf shrub fine root litter) in different FTYPEs. Models are from Ojanen et al. (2014), except for the constants of *V. myrtillus* and *V. vitis-idaea* FTYPEs, which are area weighted means of constants of their subtypes; BA = basal area (m^2 ha^{-1}).

Drained peatland forest site type	Ground vegetation litter input (g m^{-2} year^{-1})
Herb rich drained peatland forest	$227 - 4.52 \times BA$
Vaccinium myrtillus drained peatland forest	$227 - 4.52 \times BA$
Vaccinium vitis-idaea drained peatland forest	$256 - 4.52 \times BA$
Dwarf shrub drained peatland forest	$298 - 4.52 \times BA$
Cladonia drained peatland forest	$187 - 4.52 \times BA$

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Table 6. Empirical regression models of CO_2 release from peat and litter decomposition for FTYPEs as used in the new method. The regression models are from Ojanen et al. (2014), except that the constants for the *V. myrtillus* and *V. vitis-idaea* FTYPEs are
780 **weighted means of constants of the two subtypes; BA = tree stand basal area (m^2 ha^{-1}), T = mean May–October air temperature (°C).**

Drained peatland forest site type	Decomposition (g CO_2 m^{-2} year^{-1})
Herb rich drained peatland forest	$-1383 + 14.74 \times BA + 242.8 \times T$
Vaccinium myrtillus drained peatland forest	$-1440 + 14.74 \times BA + 242.8 \times T$
Vaccinium vitis-idaea drained peatland forest	$-1662 + 14.74 \times BA + 242.8 \times T$
Dwarf shrub drained peatland forest	$-1771 + 14.74 \times BA + 242.8 \times T$
Cladonia drained peatland forest	$-1814 + 14.74 \times BA + 242.8 \times T$

785

Table for Appendix A

Table A1. NFI12 estimates of the areas of drained peatland forest site types in Southern and Northern Finland
790 **together with their standard errors (s.e.) and relative standard errors (RSE) due to sampling assessed as explained in**
Korhonen et al. (2021, Supplementary file S1).

Region	Drained peatland forest site type	Area, Mha	s.e., Mha	RSE, %
Southern Finland	Herb rich type (Rhtkg)	0.336	0.012	3.5
	Vaccinium myrtillus type (Mtkg)	0.750	0.018	2.4
	Vaccinium vitis-idaea type (Ptkg)	0.704	0.019	2.6
	Dwarf shrub type (Vatkg)	0.490	0.016	3.3
	Cladonia type (Jätkg)	0.018	0.003	14.8
Northern Finland	Herb rich type (Rhtkg)	0.173	0.011	6.1
	Vaccinium myrtillus type (Mtkg)	0.488	0.018	3.8
	Vaccinium vitis-idaea type (Ptkg)	0.972	0.027	2.8
	Dwarf shrub type (Vatkg)	0.572	0.021	3.7
	Cladonia type (Jätkg)	0.019	0.003	16.8

Appendix A. Assessment of uncertainty

Uncertainty assessments were developed for all annual estimates of soil CO_2 balance in Southern Finland, Northern Finland,
and whole country, as well as for the estimates of change in balance between years 1990 and 2016. The accounted sources of
uncertainty included NFI sampling errors in estimates of the areas of drained peatland forest types and basal area and
biomass of trees, and uncertainty about the parameters of the models, litter production and turnover rates, and mean dwarf
shrub coverage.

Notation

Soil CO_2 balance, Mt year^{-1} (net emission positive), in year t over forests representing site type p in region r (Southern
Finland or Northern Finland) was estimated as $Y_{rpt} = A_{rpt} y_{rpt}$, where A_{rpt} is the interpolated NFI estimate of the area,
Mha, of site type p in region r and

$$y_{rpt} = R_{rpt}/100 - 44 \cdot 0.5(G_{rpt}/100 + F_{rpt}/100 + L_{rpt} + H_{rt})/12,$$

is the estimated net CO_2 exchange per area unit, Mg ha^{-1} year^{-1}. The components of y_{rpt} are

$$R_{rpt} = \alpha_1 B_{rpt} + \alpha_2 T_{rpt} + \beta_{1,p},$$

the prediction of peat and litter decomposition, g CO_2 m^{-2} year^{-1}, obtained using model (13) of Ojanen et al. (2014, Table
A.5),

$$G_{rpt} = \alpha_3 B_{rpt} + \beta_{2,p},$$

the biomass of litter from ground vegetation, g m^{-2} year^{-1}, as predicted by model (12) of Ojanen et al. (2014, Table A.4),

$$F_{rpt} = \alpha_4 \phi_p \left(\sum_s \delta_s B_{rpst} + \alpha_5 \mu_p + \gamma_r \right),$$

the biomass of litter from fine roots, g m^{-2} year^{-1}, as predicted by model (8) of Ojanen et al. (2014, Table A.2), corrected
according to Laiho and Finér (1996) and combined with the site-type specific turnover rates,
L_{rpt}, the biomass of litter from living trees (excluding fine roots), Mg ha^{-1} year^{-1}, and
H_{rt}, the biomass of residual organic matter from tree harvesting and natural mortality after decomposition, Mg ha^{-1} year^{-1}.
In the equations above, $B_{rpt} = \sum_s B_{rpst}$ is the basal area of trees, m^2 ha^{-1}, based on NFI, s refers to tree species group, T_{rpt}
is the time-smoothed mean May-October air temperature, °C, over sites of type p in region r, and the remaining static
components of the predictors are based on earlier studies. Model parameters

$$\boldsymbol{\theta}_R = [\alpha_1 \quad \alpha_2 \quad \beta_{1,\text{Rhtkg}} \quad \beta_{1,\text{Mtkg}} \quad \beta_{1,\text{Ptkg}} \quad \beta_{1,\text{Vatkg}} \quad \beta_{1,\text{Jätkg}}]'$$

are based on Ojanen et al. (2014, Table A.5),

820    $\boldsymbol{\theta}_G = [\alpha_3 \quad \beta_{2,\text{Rhtkg}} \quad \beta_{2,\text{Mtkg}} \quad \beta_{2,\text{Ptkg}} \quad \beta_{2,\text{Vatkg}} \quad \beta_{2,\text{Jätkg}}]'$

on Ojanen et al. (2014, Table A.4), and

   $\boldsymbol{\theta}_F = [\delta_{\text{pine}} \quad \delta_{\text{spruce}} \quad \delta_{\text{deciduous}} \quad \alpha_5 \quad \gamma_{\text{south}} \quad \gamma_{\text{north}}]'$

on Ojanen et al. (2014, Table A.2). Mean dwarf shrub coverages μ_p are from Table A.3 of Ojanen et al. (2014), correction

$\alpha_4 = 1.043$ is based on Laiho and Finér (1996), and turnover rates ϕ_p are reported in this article (Table 3). $q = 5$ is the

825 number of peatland forest site types, and the applied values of the site-type specific parameters are listed in Tables 3-6. For

$p \in \{2,3\}$, the values $\beta_{1,p}, \beta_{2,p}, \mu_p$ are area-weighted averages of the values in Ojanen et al. (2014, Tables A.5, A.4, and A.3)

over the two subtypes I and II ($w_{\text{MtkgI}} = 0.615$, $w_{\text{PtkgI}} = 0.6$).

The region-specific annual estimates were $Y_{rt} = \sum_p Y_{rpt}$, country-wide annual estimates, $Y_t = \sum_r Y_{rt}$, and corresponding

change estimates, $Y_{r,2016} - Y_{r,1990}$ and $Y_{2016} - Y_{1990}$.

830 **Uncertainty components**

Uncertainty in estimates A_{rpt} and B_{rpst} due to NFI sampling was based on NFI12 (Tables A1 and A2) relying on the

assumptions that (i) the relative standard errors (RSE) $\text{Var}^{1/2}(A_{rpt})/A_{rpt}$ and $\text{Var}^{1/2}(B_{rpt})/B_{rpt}$ are unchanged across time,

i.e., same and equal to the corresponding RSE's in Tables A1 and A2 for all t and (ii) the estimates $A_{rp,1990}$ and $A_{rp,2016}$ are

uncorrelated, as well as $B_{rps,1990}$ and $B_{rps,2016}$. Assumption (i) is justified by the relatively constant sampling effort of NFI

835 across time, and assumption (ii) by the fact that the estimates for the two years are completely based on different NFI

campaigns.

Uncertainty due to estimation of parameters in models (8), (12), and (13) of Ojanen et al. (2014) was based on covariance

matrices Σ_R, Σ_G, and Σ_F of parameter vectors $\boldsymbol{\theta}_R, \boldsymbol{\theta}_G$, and $\boldsymbol{\theta}_F$ (Tables A3 - A5) derived from Tables A.5, A.4, and A.2 of

Ojanen et al. (2014). In particular,

840    $\text{Var}(\beta_{1,\text{Mtkg}}) = w_{\text{MtkgI}}^2 \text{Var}(\beta_{\text{MtkgI}}) + w_{\text{MtkgII}}^2 \text{Var}(\beta_{\text{MtkgII}}) + 2w_{\text{MtkgI}}w_{\text{MtkgII}}\text{Cov}(\beta_{\text{MtkgI}}, \beta_{\text{MtkgII}})$, and so on.

Variances of site-type specific estimates of dwarf shrub coverage, $\text{Var}(\mu_p)$, were similarly derived from those presented in

Table A.3 of Ojanen et al. (2014) and variances of fine root turnover rates, $\text{Var}(\phi_p)$ were based on expert judgement (Table

A6), as was that of the deep-root correction α_4, $\text{Var}(\alpha_4) = 0.012^2$.

Relative standard errors of litter production estimates from living trees on drained peatland sites were estimated by region

845 from NFI8 and NFI11 as explained in Lehtonen and Heikkinen (2016). For annual results, RSE's from NFI11 were applied

to the litter series L_{rpt} across all site types and times (Table A7). When aggregating to the whole country and assessing the

uncertainty in change between years 1990 and 2016, correlations between regions and time points (Table A8), due to

applying the same biomass models and litter production rates, were also taken into consideration.

Relative standard error of litter originating from harvests and natural mortality was estimated similarly from NFI11 and was

850 applied to the residual series H_{rt}.

Apart from correlations discussed above, all sources of uncertainty were assumed mutually uncorrelated.

Impacts of uncertainty components on soil CO_2 balance estimates

Assuming negligible correlation between area estimates A_{rpt}, variances of region-specific annual balance estimates $Y_{rt} = \sum_p A_{rpt} y_{rpt}$ due to NFI sampling variance in area estimates A_{rpt} were obtained as

855 $$\text{Var}_A(Y_{rt}) = \sum_p y_{rpt}^2 \text{Var}(A_{rpt}),$$

variances of country-wide balanced estimates as

$$\text{Var}_A(Y_t) = \sum_{r,p} y_{rpt}^2 \text{Var}(A_{rpt}),$$

and variances of the change in balance between years 1990 and 2016 as

$$\text{Var}_A(Y_{r,2016} - Y_{r,1990}) = \sum_p [y_{rp,2016}^2 \text{Var}(A_{rp,2016}) + y_{rp,1990}^2 \text{Var}(A_{rp,1990})],$$

860 and

$$\text{Var}_A(Y_{2016} - Y_{1990}) = \sum_{r,p} [y_{rp,2016}^2 \text{Var}(A_{rp,2016}) + y_{rp,1990}^2 \text{Var}(A_{rp,1990})].$$

To evaluate the impact of NFI sampling variance in basal area estimates B_{rpst}, balance estimates were rewritten as

$$Y_{rt} = \sum_p C_{1,rpt} + \sum_{p,s} C_{2,rpst} B_{rpst},$$

where intercepts

865 $$C_{1,rpt} = A_{rpt} \left\{ \frac{\alpha_2 T_{rpt} + \beta_{1,p}}{100} - \frac{44 \cdot 0.5}{12} \left[\frac{\beta_{2,p} + \alpha_4 \phi_p (\alpha_5 \mu_p + \gamma_r)}{100} + L_{rpt} + H_{rt} \right] \right\}$$

do not depend on basal area estimates and

$$C_{2,rpst} = A_{rpt} \left\{ \frac{\alpha_1}{100} - \frac{44 \cdot 0.5}{12} \left[\frac{\alpha_3 + \alpha_4 \phi_p \delta_s}{100} \right] \right\}.$$

Then, for the annual region-specific balance estimates,

$$\text{Var}_B(Y_{rt}) = \sum_{p,s} C_{2,rpst}^2 \text{Var}(B_{rpst}),$$

870 and the variances for the whole country and for the change are obtained similarly (cf. area estimates above).

Considering uncertainty in the parameters of the peat and litter decomposition model, the balance estimates can also be expressed as

$$Y_{rt} = C_{3,rt} + C_{4,rt}\alpha_1 + C_{5,rt}\alpha_2 - \sum_p C_{6,rpt}\beta_{1,p},$$

where $C_{3,r}$ does not depend on these parameters,

875 $$C_{4,rt} = \sum_p \frac{A_{rpt}B_{rpt}}{100},$$

$$C_{5,rt} = \sum_p \frac{A_{rpt}T_{rpt}}{100},$$

and

$$C_{6,rpt} = A_{rpt}/100.$$

Thus, the impact of uncertainty in the parameters of the peat and litter decomposition model is

880 $$\text{Var}_R(Y_{rt}) = \mathbf{c}_{rt}'\Sigma_R\mathbf{c}_{rt},$$

where $\mathbf{c}_{rt} = (C_{4,rt}, C_{5,rt}, C_{6,r,\text{Rhtkg},t}, \ldots, C_{6,r,\text{Jatkg},t})'$. The corresponding variances for the whole country were obtained using coefficients

$$C_{4,t} = \sum_{r,p} \frac{A_{rpt}B_{rpt}}{100}, \quad C_{5,t} = \sum_{r,p} \frac{A_{rpt}T_{rpt}}{100}, \text{ and } C_{6,pt} = \sum_r \frac{A_{rpt}}{100}$$

and for the region-specific change estimates with coefficients

885 $$C_{4,r} = \sum_p \frac{A_{rp,2016}B_{rp,2016} - A_{rp,1990}B_{rp,1990}}{100} \text{ etc.}$$

Variances $\text{Var}_G(Y)$ and $\text{Var}_F(Y)$ due to parameter uncertainty in ground vegetation and fine-root litter production model were derived from covariance matrices Σ_G and Σ_F and variances $\text{Var}(\alpha_4)$, $\text{Var}(\phi_p)$, $\text{Var}(\mu_p)$ in a similar manner.

Variances in region-specific annual balance estimates due to uncertainty in litter input from living trees were estimated as

$$\text{Var}_L(Y_{rt}) = \sum_p \left(A_{rpt}\frac{44 \cdot 0.5}{12}\right)^2 \text{Var}(L_{rpt}),$$

890 where variances $\text{Var}(L_{rpt})$ were based on RSE's in Table A7. When propagating these to the whole country, covariances $\text{Cov}(L_{\text{south},pt}, L_{\text{north},pt})$, based on the NFI11 correlation 0.539 (Table A8), were accounted for, as were covariances between time points when considering the change estimates. For instance,

$$\text{Var}_L(Y_{2016} - Y_{1990}) = \mathbf{c}'\Sigma\mathbf{c},$$

where $\mathbf{c} = (1,1,-1,-1)'$ and Σ is the covariance matrix of $(Y_{\text{south},2016}, Y_{\text{north},2016}, Y_{\text{south},1990}, Y_{\text{south},1990})'$ derived using the

895   correlation matrix of Table A8 and variances based on NFI11 for $Y_{r,2016}$ and NFI8 for $Y_{r,1990}$.

Finally, variances due to uncertainty in litter input from harvests and natural mortality,

$$\text{Var}_H(Y_{rt}) = \left(\frac{44 \cdot 0.5}{12}\sum_p A_{rpt}\right)^2 \text{Var}(H_{rt}),$$

were based on RSE's in Table A7 and on the assumption that the input estimates are uncorrelated across regions and time points. To combine all sources of uncertainty, source-specific variances were simply added up:

900   $$\text{Var}(Y) = \text{Var}_A(Y) + \text{Var}_B(Y) + \text{Var}_R(Y) + \text{Var}_G(Y) + \text{Var}_F(Y) + \text{Var}_L(Y) + \text{Var}_H(Y).$$

Following IPCC Guidelines (Frey et al. 2006), percentage uncertainties were defined as

$$U_Y = 100 \times 1.96\sqrt{\text{Var}(Y)}.$$

Results

Uncertainty of year 2016 (positive) balance estimates for Southern Finland and whole country are less than 100% (Table
905   A9), which indicates that the soils of drained organic peatlands are statistically significant sources of CO_2 according to our model predictions. However, the 2016 balance in Northern Finland does not differ significantly from zero. Analogous results were also obtained for the change in emission between years 1990 and 2016 (Table A10).

The greatest impact was due to uncertainty in the parameter estimates of the peat and litter decomposition model ($38 - 70$ % of variance in 2016 balance estimates, $68 - 74$ % in change estimates; Tables A9 and A10). The impact of uncertainty in
910   fine-root model parameters was relatively modest ($7 - 15$ %) in change estimation, but somewhat greater ($17 - 4$ 2%) in annual estimates. From other sources of uncertainty, only litter input from living trees exceeded 10% impact in any of the estimates. In particular, uncertainty in NFI estimates of site type areas and basal areas of trees had negligible impact on annual balance estimates and contributed less than 5 % of the variance of change estimates.

The errors in annual balance estimates are positively correlated, because the same model parameters with same estimation
915   errors were applied throughout the series. For example,

$$\text{Corr}(Y_{2016}, Y_{1990}) = \frac{\text{Cov}(Y_{2016}, Y_{1990})}{\sqrt{\text{Var}(Y_{2016})\text{Var}(Y_{1990})}} = \frac{\text{Var}(Y_{2016}) + \text{Var}(Y_{1990}) - \text{Var}(Y_{2016} - Y_{1990})}{2\sqrt{\text{Var}(Y_{2016})\text{Var}(Y_{1990})}} = 0.92.$$

Due to these correlations, change estimates can differ significantly from zero even if confidence intervals of annual estimates have a large overlap.

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