



The current greenhouse gas impact of forestry-drained boreal peatlands

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ABSTRACT

We estimated the soil CO₂ balance of 68 forestry-drained boreal peatland sites in Finland by subtracting the litter input to soil from the CO₂ efflux from soil. We also measured soil–atmosphere fluxes of CH₄ and N₂O and the CO₂ sink of the growing tree stand in order to assess the current greenhouse gas impact of the study sites.

The soil was, on average, a CO₂ source of $+190 \pm 70 \text{ g m}^{-2} \text{ year}^{-1}$ at the fertile *Herb-rich* and *Vaccinium myrtillus* type sites, but a CO₂ sink of $-70 \pm 30 \text{ g m}^{-2} \text{ year}^{-1}$ at the poor *Vaccinium vitis-idaea* and *Dwarf shrub* type sites. The source increased at the fertile and the sink decreased at the poor sites as the water table deepened. The source at the fertile sites also increased by increasing temperature sum, the highest CO₂ sources being around $+1000 \text{ g m}^{-2} \text{ year}^{-1}$ at well drained sites in Southern Finland.

Both fertile and poor sites had a climate cooling impact. The sink in CO₂ equivalents at the fertile sites was $-690 \pm 90 \text{ g m}^{-2} \text{ year}^{-1}$ and at the poor sites $-540 \pm 70 \text{ g m}^{-2} \text{ year}^{-1}$. The greater sink at the fertile sites was due to clearly better tree growth, their tree stand CO₂ sink being $-880 \pm 60 \text{ g m}^{-2} \text{ year}^{-1}$ compared to the $-490 \pm 60 \text{ g m}^{-2} \text{ year}^{-1}$ at the poor sites.

Ditching-based forestry can be climatically sustainable at nutrient-poor boreal peatlands since the peat soil continues to be a CO₂ sink even after drainage. At the fertile sites, forestry will inevitably lead to loss of carbon in the long term, unless the tree biomass is stored after cuttings, for example in wooden buildings or as biochar in agricultural soils.

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1. Introduction

Pristine boreal peatlands are sinks of atmospheric carbon dioxide (CO₂) as carbon (C) is bound into the accumulating peat (e.g., Turunen et al., 2002; Aurela et al., 2007; Nilsson et al., 2008; Sagerfors et al., 2008). At the same time, methane (CH₄) is released due to the wet, anoxic conditions in the soil (e.g., Waddington and Roulet, 2000; Rinne et al., 2007; Leppälä et al.,

2011). Although the nitrogen content of peat can be high, nitrous oxide (N₂O) emissions are generally low, occurring mainly at the most fertile sites (Regina et al., 1996; Drewer et al., 2010; Lohila et al., 2010). Across the decadal to centennial time scales, a pristine boreal peatland may have a climate cooling or warming impact depending on the balance between the CO₂ sink and the CH₄ source (Frolking et al., 2006; Drewer et al., 2010). Over the millennial time scale, the climate impact will inexorably be cooling due to the constant peat accumulation (Frolking et al., 2006; Frolking and Roulet, 2007).

Drainage for forestry may have both climate cooling and climate warming impacts: When the groundwater table is lowered, CH₄ emissions reduce, cease or even reverse (Maljanen et al., 2010; Ojanen et al., 2010). With well functioning drainage and an adequate nutrient supply, tree stand biomass starts to increase, which results in a considerable CO₂ sink (Tomppo, 1999; Minkkinen et al., 2001). On the other hand, if the peat layer starts to degrade, as is the case after drainage for agriculture (Maljanen et al., 2007), the soil turns into a source of CO₂. In agricultural peat soils, N₂O emissions can also be high due to N fertilisation, and may remain at a high level even decades after possible abandonment or afforestation (Regina et al., 2004; Maljanen et al., 2007, 2010). In

Abbreviations: BA, tree stand basal area; BM, biomass; BM_{cr}, biomass of tree coarse roots; BM_{ds}, aboveground biomass of dwarf shrubs; BM_{dsr}, biomass of dwarf shrub rhizomes; BM_{fr}, biomass of tree and dwarf shrub fine roots; BM_h, aboveground biomass of herbs other than sedges and grasses; BM_{sg}, aboveground biomass of sedges and grasses; D, decomposition; D_{litter}, decomposition of the fresh litter layer; L, litter production; L_m, litter production of mosses; NE_{CH₄}, soil net CH₄ exchange; NE_{CO₂ soil}, soil net CO₂ exchange; NE_{CO₂ trees}, tree stand net CO₂ exchange; NE_{N₂O}, soil net N₂O exchange; PC, projection coverage; PC_{ds}, projection coverage of dwarf shrubs; PC_h, projection coverage of herbs other than sedges and grasses; PC_m, projection coverage of mosses; PC_{sg}, projection coverage of sedges and grasses; R_{het}, heterotrophic soil respiration; R_{ref}, R_{het} at reference temperature; t, turnover rate; T_{MO}, average May–October air temperature; WT, average frost-free season water table.

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forestry-drained peatlands, N₂O emissions are generally low although high emissions may occur at fertile sites (Maljanen et al., 2010; Ojanen et al., 2010).

Tree growth has been widely studied, and the estimation of its CO₂ sink is well established (Statistics Finland, 2011; Swedish Environmental Protection Agency, 2011). Data on CH₄ and N₂O fluxes in forestry-drained boreal peatlands are also adequately available (e.g., von Arnold et al., 2005a, 2005b; Maljanen et al., 2010; Ojanen et al., 2010). On the other hand, only a few publications deal with direct CO₂ balance measurements on drained, forested peatlands (Hargreaves et al., 2003; Lohila et al., 2007, 2011). Long-term data on changes in soil C storage after forestry drainage are even less common (Minkkinen and Laine, 1998; Minkkinen et al., 1999).

The scarcity of results on soil C balance in forested peatlands is obviously due to three issues: (1) Soil C balance cannot be estimated in treed sites by the widely used transparent chamber method (e.g., Alm et al., 2007), as soil CO₂ exchange and C balance do not equal due to the litter production and belowground C allocation of the tree stand (Ojanen et al., 2012). (2) Ecosystem CO₂ exchange measurements by eddy covariance technique are expensive and laborious and, thus, inadequate for extensive studies. (3) Repeated sampling of soil C stocks is difficult as alterations in soil water level and in C stocks change the elevation of the soil surface. The changes in soil C stocks are also very small compared to the storage.

The results of the sole extensive empirical study on soil C balance, addressing subsidence and the change in peat bulk density, suggest that peatlands poor in nutrients accumulate peat even after forestry drainage (Minkkinen and Laine, 1998). This conclusion is supported by the results of an eddy covariance case study on a nutrient-poor peatland in Southern Finland (Lohila et al., 2011). These results contrast with the view that peatland drainage inevitably leads to C losses (e.g., Couwenberg, 2011). Thus, more empirical data are needed to verify or challenge these results. Furthermore, not only the soil C balance but also the CH₄ and N₂O balance as well as the tree stand CO₂ balance affect the climate, hence they should likewise be included when estimating the current greenhouse gas (GHG) impact of forestry-drained peatlands.

We aim at studying the soil net CO₂ exchange in boreal forestry-drained peatlands and unravelling its controls by employing an approach that combines well-established methods of assessing soil–atmosphere gas exchange and C inputs to soil in a new way. We also estimate the current greenhouse gas impact of forestry-drained peatlands, including both tree stand and soil while taking into account all the three relevant GHGs. For these purposes, we estimated the net exchange of CO₂, CH₄, and N₂O for 68 peatland sites in Finland by measuring soil–atmosphere fluxes of GHGs and estimating litter production and tree stand biomass increment. Through our extensive empirical data set and the approach that addresses several relevant factors, we will discuss the ecological conditions in which forestry-drained boreal peatland sites may function as GHG sinks or sources.

2. Materials and methods

2.1. Study sites

The 68 study sites covered the span of the south and middle boreal vegetation zones in Finland (see Fig. 1 in Ojanen et al., 2010). Sites belonging to each drained peatland site type (Laine, 1989; see Vasander and Laine, 2008) were equally included to represent the continuum from the most fertile *Herb-rich* type via *Vaccinium myrtillus* types I and II and *Vaccinium vitis-idaea* types I and II to the poor *Dwarf shrub* type. Types with suffix I include sites,

which before drainage were forested mires; types with suffix II include sites, which before drainage were sparsely treed or treeless mires. All the sites had been drained at least 20 years prior to the study.

The study sites were selected from the network of 3009 permanent sample plots connected to the 8th Finnish National Forest Inventory. Plot-wise vegetation surveys (Reinikainen et al., 2000; Mäkipää and Heikkinen, 2003) were carried out in these plots in 1995, and the data on species-specific projection coverages from those surveys were used in this study for estimating the aboveground biomass and litter production of ground vegetation.

2.2. Soil net CO₂ exchange

The soil net CO₂ exchange (NE_{CO₂ soil}) was estimated by subtracting the rate of CO₂ production due to the decomposition of organic matter (*D*) from the rate of litter production (*L*) as (Ojanen et al., 2012):

$$NE_{CO_2 \text{ soil}} = L - D \quad (1)$$

L was estimated as the annual sum of aboveground and belowground litter production of vascular plants and mosses. *D* was estimated as the annual sum of measured heterotrophic soil respiration (*R*_{het}; Ojanen et al., 2010), adjusted to correspond to the 2001–2010 average summer air temperature, plus modelled decomposition of the fresh litter layer (*D*_{litter}). Fifty percent of organic matter dry mass was considered to be carbon in all calculations (Vávřová et al., 2009; Straková et al., 2010).

2.3. Current greenhouse gas impact

The current greenhouse gas impact describes the site's annual net effect on the climate. It was estimated as the sum of the ecosystem–atmosphere net exchange (NE) of CO₂, N₂O and CH₄ in CO₂ equivalents (GWP₁₀₀; IPCC, 2007):

$$\text{Greenhouse gas impact} = NE_{CO_2 \text{ soil}} + NE_{CO_2 \text{ trees}} + 25 NE_{CH_4} + 298 NE_{N_2O} \quad (2)$$

For CO₂, NE consisted of NE_{CO₂ soil} and the current CO₂ sink of the growing tree stand (NE_{CO₂ trees}). NE_{CO₂ trees} was estimated as the difference of living tree biomass between two consecutive assessments, spanning a 5–13-year period, depending on the study site. The C stocks in ground vegetation are very small compared to those of the soil and tree stand. The short-term changes in those C stocks are even smaller, if any (Laiho et al., 2003), and were therefore omitted from the calculations. For CH₄ and N₂O, the NEs were those annual soil–atmosphere fluxes estimated by Ojanen et al. (2010). Positive values indicate sources (climate warming impact) and negative values sinks (climate cooling impact) of GHGs.

2.4. Gas flux and water table measurements

Soil–atmosphere GHG fluxes were measured in 2007–2008 at each site (Ojanen et al., 2010). For the measurement of *R*_{het}, five circular points (*d*, 400 mm) were trenched with 30 cm deep metal cylinders in autumn 2006. After trenching, ground vegetation and the layer of fresh, loose litter were removed. All the shed aboveground litter was also collected from the points each time before gas flux measurement. The removal of litter facilitated keeping the points moss-free and also provided an estimate of the tree stand aboveground litter production.

*R*_{het} was measured every 2–3 weeks over the May–October period using a portable infrared gas analyzer (EGM-4, PP-Systems) equipped with an opaque closed chamber (*d*, 315 mm, *h*,

149 mm). Annual fluxes were calculated using nonlinear regressions between soil temperature and measured, momentary R_{het} .

Gas samples for calculating CH_4 and N_2O fluxes were taken once a month from four intact points at each site using an opaque closed chamber (d , 315 mm, h , 295 mm) equipped with a fan. Samples were analysed with a gas chromatograph using FI and EC detectors (see Mäkiranta et al., 2012 for details). Annual fluxes were interpolated from measurements.

Depth to the water table from the peat surface was measured from a perforated tube in the peat soil each time that R_{het} was measured. The mean of the measurements for each site was calculated to describe the average frost-free season water table (WT). Negative values indicate WT below soil surface.

2.5. Decomposition, D

R_{het} was adjusted to account for the average May–October air temperature (T_{MO} , °C) during the period 2001–2010. This was done by applying the regression (r^2 , 0.95) of R_{het} with T_{MO} and the R_{het} at reference temperature (R_{ref} , $\text{g m}^{-2} \text{h}^{-1}$ of CO_2) to the difference between the T_{MO} (Venäläinen et al., 2005) in 2007–2008 and in 2001–2010 of each site (Ojanen et al., 2010):

$$R_{\text{het}} = -1090 + 4910 R_{\text{ref}} + 110 T_{\text{MO}} \quad (3)$$

As the measured R_{het} did not include the CO_2 efflux from D_{litter} , its contribution was estimated using Yasso07 litter decomposition model (Tuomi et al., 2011). The model describes the aerobic decomposition of litter as a function of litter chemical composition, air temperature and precipitation.

As input to the model, we used the annual aboveground litter production estimated in this study and average monthly weather data for the years 2001–2010 estimated for each site from the database of the Finnish Meteorological Institute (Venäläinen et al., 2005). The chemical compositions for the different litter types were adopted from Vávřová et al. (2009) and Straková et al. (2010).

We ran the model for each site as many years as needed (on average 2 years) to equal the carbon pool of the modelled litter layer to that of the removed litter layer (range 10–800 g m^{-2} of C). Then we ran the model one more year to account for the decomposition of the newly shed litter that was removed from the R_{het} points during the monitoring period. The CO_2 efflux from D_{litter} estimated for that model year was then added to R_{het} to estimate D :

$$D = R_{\text{het}} + D_{\text{litter}} \quad (4)$$

2.6. Litter production, L

2.6.1. Aboveground

Aboveground litter of trees was collected from the five R_{het} measurement points at each site. Litter collection started after setting up the monitoring points in autumn 2006 and continued for 4 years at 28 sites and for 2 years at the rest of the sites. Litter was dried at 40 °C to constant mass and weighed. Subsamples were further dried at 105 °C for determining the remaining moisture content. The samples from 23 sites were sorted out to estimate the average proportion of particles not considered tree litter (vascular ground vegetation litter and mosses). This proportion, 6%, was reduced from the dry mass of tree litter in all sites.

As ground vegetation was removed when setting up the R_{het} measurement points, its litter production could not be estimated from the collected litter. Litter production (L) of vascular ground vegetation was estimated based on biomasses (BMs) and turnover rates (t , year^{-1} ; Table 1):

$$L = t \text{ BM} \quad (5)$$

Species-specific aboveground BMs for the vascular ground vegetation at 101 forestry-drained peatland sites were estimated in the BioSoil project in 2006 (Ilvesniemi et al., 2006, 2008). All BMs were sampled from five 30 × 30 cm squares at each site during July–August and corresponding projection coverages (PCs) were visually estimated. Fourteen of our study sites were BioSoil sites and thus had BMs measured. To estimate the BMs for the rest of our sites, we used the 101 BioSoil sites to estimate ratios between PCs ($\text{m}^2 \text{m}^{-2}$) and BMs (g m^{-2} of dry mass) of dwarf shrubs (ds), sedges and grasses (sg), and other herbs (h) as follows. Parameter standard errors are depicted in parentheses.

$$\text{BM}_{\text{ds}} = 4.2 (0.2) \text{ PC}_{\text{ds}} \quad (6)$$

$$\text{BM}_{\text{sg}} = 4.3 (0.5) \text{ PC}_{\text{sg}} \quad (7)$$

$$\text{BM}_{\text{h}} = 0.68 (0.06) \text{ PC}_{\text{h}} \quad (8)$$

Using those ratios, we estimated the aboveground BMs for each of our non-BioSoil study sites by utilising the PCs estimated in the vegetation surveys in 1995 on the very same plots.

Similarly, the litter production of mosses (L_{m} , $\text{g m}^{-2} \text{year}^{-1}$ of dry mass) was estimated by applying the ratio between L_{m} and the PC of mosses (PC_{m}) estimated in 1995:

$$L_{\text{m}} = 1.6 (0.3) \text{ PC}_{\text{m}} \quad (9)$$

This ratio was based on a study on one *V. myrtillus* type and one Dwarf shrub type drained peatland in Southern Finland (Laiho et al., 2011) and unpublished data from one Dwarf shrub type site in Southern Finland (for site description, see Badorek et al., 2011).

2.6.2. Belowground

Five belowground biomass samples (roots and rhizomes) per site were taken from 31 of the study sites in August–November, partly in 2007 and partly in 2008. The area of one sample was 15 × 15 cm and it reached to 20 cm depth from the soil surface. Biomass was separated according to species group and diameter (d) into four compartments: coarse tree roots ($d > 2$ mm), dwarf shrub rhizomes ($d > 2$ mm), tree and dwarf shrub fine roots ($d \leq 2$ mm), and herbaceous roots. This separation was considered to divide the below ground biomass into compartments that have distinctly different turnover rates (Table 1).

According to Laiho and Finér (1996), 4.3% of tree and dwarf shrub fine roots and 4.9% of coarse roots and rhizomes are located deeper than 20 cm in boreal forestry-drained peatlands. Our measured values were corrected accordingly to account for the whole soil profile.

Measured biomasses were generalised to the rest of the sites using ratios of tree coarse root biomass (BM_{cr} , g m^{-2} of dry mass) to tree stand basal area (BA , $\text{m}^2 \text{m}^{-2}$) and dwarf shrub rhizomes (BM_{dsr} , g m^{-2}) to dwarf shrub projection coverage (PC_{ds}). Tree and dwarf shrub fine root biomass (BM_{fr} , g m^{-2}) was generalised with a linear regression between biomass and BA. A constant had to be included as there still are dwarf shrub roots when the tree stand BA reaches zero. Parameter standard errors are in parentheses.

Table 1
Biomass turnover ratios (year^{-1}) used in the estimation of litter production.

Component	Value	Source
<i>Aboveground</i>		
All herbaceous	1	Biomass renews annually
Dwarf shrubs	0.15	Unpublished data (Badorek et al.)
<i>Belowground</i>		
Tree roots $d > 2$ mm	0.015	Liski et al. (2006)
Dwarf shrub rhizomes	0.08	Finér and Laine (1998)
Arboreal roots < 2 mm	0.85	Liski et al. (2006)
Herb roots	1.25	Laiho et al. (2003)

$$BM_{cr} = 49 \text{ (5) BA} \quad (10)$$

$$BM_{dsr} = 9.8 \text{ (1.4) PC}_{ds} \quad (11)$$

$$BM_{fr} = 110 \text{ (60) + 12 (3) BA} \quad (12)$$

For the herbaceous roots, no correlation was found, and thus their biomass was generalised simply by taking the mean of the measured values, 37 (10) g m⁻² of dry mass. Belowground litter production was then estimated by applying turnover ratios by compartment (Eq. (5), Table 1).

2.7. Tree stand CO₂ sink

The living tree stand was measured at each site in autumn 1995 and again in spring 2006 (the 14 BioSoil sites) or autumn 2008 (the other study sites) using circular sample plots with a radius of 3.09–25.24 m, depending on tree diameter at breast height (D_{bh}). Tree height and crown length were measured from sample trees. In 2006 and 2008, the sample trees were measured also for annual D_{bh} increment during the last 20 years.

After generalising the sample tree characteristics to all trees via regressions against D_{bh} , D_{bh} distributions (trees ha⁻¹) including an estimate for tree height and crown length in each 1-cm D_{bh} class were created for the years 1995, 2000 and 2005/2008. Species-specific single-tree biomass models of Repola (2008, 2009) and Laiho and Finér (1996) were then applied to the distributions, and living tree stand biomasses were calculated for each site. The procedure is described in more detail by Ojanen et al. (2012).

Average annual tree stand CO₂ sink *ex post facto* for the living tree stands was calculated as (biomass 2008 – biomass 2000)/8 (50 sites) or (biomass 2005 – biomass 2000)/5 (11 sites). Comparison between tree data in 1995 and 2006/2008 showed that tree mortality was negligible. It was thus neglected in the calculations. Increment measurements were not available for seven sites. At four of those sites, no cuttings had occurred between 1995 and 2006/2008 and the tree stand CO₂ sink could be calculated as (biomass 2008 – biomass 1995)/13 (3 sites) or (biomass 2005 – biomass 1995)/10 (1 site). For three sites, the tree stand CO₂ sink could not be estimated because of heavy cuttings between the stand measurements.

3. Results

3.1. Soil net CO₂ exchange

Trees dominated the aboveground litter production, and fine roots of trees and dwarf shrubs dominated the belowground litter production (L) (Table 2). Herbaceous plants comprised on average 12% and mosses 10% of L .

Measured R_{het} was the main component (83%) of decomposition (D), but also the modelled fresh litter layer decomposition represented a considerable share, 15% (Table 3). The adjustment of R_{het} to the average temperature of 2001–2010 increased D by only 2%, as T_{MO} was only 0.24 °C higher in 2001–2010 compared to 2007–2008.

The three most fertile site types were, on average, CO₂ sources and the three poorest site types were, on average, CO₂ sinks (Table 4). Both D and L decreased towards poor site types, and the CO₂ source turning into a CO₂ sink was a result of D decreasing more than L . A firm correlation between L and D was observed (Fig. 1). Especially at the poor site types, the ratio was almost one to one. In the high end of the fertile sites, the correlation between L and D was loose, D being up to 1000 g m⁻² year⁻¹ of CO₂ higher than L .

Site type division was statistically significant (ANOVA, p 0.02) in explaining $NE_{CO_2 \text{ soil}}$. However, post hoc pairwise comparisons revealed that only the difference between *V. myrtillus* type I and *V. vitis-idaea* type I was significant (p < 0.05). Hence, the site types

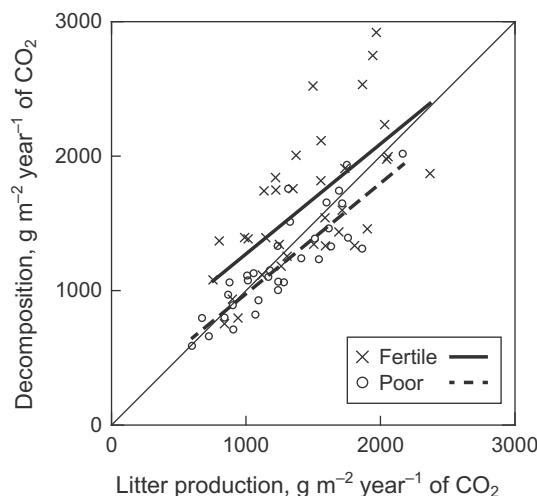


Fig. 1. Annual litter production and decomposition for each of the 68 study sites. Sites are divided into two groups: fertile, including *Herb-rich* and *Vaccinium myrtillus* type sites, and poor, including *Vaccinium vitis-idaea* and *Dwarf shrub* type sites. Lines depict linear regression.

were grouped as either 'fertile' (*Herb-rich* and *V. myrtillus* types) or 'poor' (*V. vitis-idaea* and *Dwarf shrub* types). This division was statistically significant (p , 0.001), and it explained 15% of the variation (squared multiple r) in $NE_{CO_2 \text{ soil}}$. The mean $NE_{CO_2 \text{ soil}}$ (\pm standard error) was $+190 \pm 70$ g m⁻² year⁻¹ of CO₂ for the fertile sites and -70 ± 30 g m⁻² year⁻¹ of CO₂ for the poor sites.

The soil CO₂ source at the fertile sites increased towards warmer conditions (Fig. 2). There was only one exception to this rule: an unsuccessfully drained site in Southern Finland, almost in pristine condition (WT –13 cm), was a high CO₂ sink (Fig. 2, bold). In contrast, the CO₂ sink at the poor sites showed no correlation with temperature sum.

The lowering of WT increased the CO₂ source at the fertile sites until –60 cm, after which further lowering decreased it (Fig. 3). This was caused by decreased D but unchanged L with the lowest WTs.

At the poor sites, lowering of WT decreased the sink until the lowest WT of –60 cm (Fig. 3). As there was no correlation between $NE_{CO_2 \text{ soil}}$ and temperature sum, $NE_{CO_2 \text{ soil}}$ at poor sites could be modelled by applying a simple linear regression:

$$NE_{CO_2 \text{ soil}} = -230 \text{ (80)} - 5.5 \text{ (2.7) WT} \quad (13)$$

Table 2

Mean, minimum and maximum of site-wise component and total litter production (g m⁻² year⁻¹ of CO₂). Also the mean share (%) of each component of the total litter production.

Litter component	Mean	Share	Min	Max
<i>Aboveground</i>				
Tree stand	417	30	23	1099
Dwarf shrubs	23	2	0	97
Sedges and grasses	65	5	0	518
Other herbaceous	7	1	0	54
Mosses	134	10	7	288
Sum aboveground	646	47	113	1319
<i>Belowground</i>				
Tree coarse roots $d > 2$ mm	29	2	3	86
Dwarf shrub rhizomes	33	2	0	117
Tree and dwarf shrub fine roots $d \leq 2$ mm	576	42	137	1172
Herbaceous roots	85	6	0	431
Sum belowground	723	53	240	1264
Sum all	1369	100	603	2368

Table 3

Mean, minimum, and maximum of components of organic matter decomposition (D , $\text{g m}^{-2} \text{ year}^{-1}$ of CO_2); measured heterotrophic soil respiration (R_{het}), adjustment of R_{het} to mean summer air temperature 2001–2010, and modelled litter layer decomposition (D_{litter}). Also see the mean share (%) of each component.

Component	Mean	Share	Min	Max
Measured R_{het}	1196	83	534	2455
R_{het} adjustment	27	2	–5	54
D_{litter}	211	15	26	550
D	1434	100	584	2920

This regression explained 12% of the variation (squared multiple r) in $\text{NE}_{\text{CO}_2 \text{ soil}}$ at the poor sites. It was statistically significant ($p = 0.049$). Values in parentheses represent the standard errors of the parameters.

A fertile site with both high temperature sum and efficient drainage seemed to be the prerequisite for a high CO_2 source. The four fertile sites with the highest CO_2 source had both high temperature sum (around 1300 dd.) and moderate or deep WT (–30 to –70 cm). Also, the well drained fertile sites that were not located in the area of highest temperature sums were not among the highest sources.

Attempts to fit a regression model for $\text{NE}_{\text{CO}_2 \text{ soil}}$ at the fertile sites with WT and temperature sum as independent variables failed. There were not enough data points considering the very high variation both in $\text{NE}_{\text{CO}_2 \text{ soil}}$ and WT within the fertile group.

In our entire data set, soil CO_2 source displayed an increasing trend with increasing soil bulk density and decreasing soil carbon to nitrogen ratio (data not shown). This was likely due to higher bulk density and lower CN ratio, on the average, at fertile sites compared to poor sites, as no trends were found when analysing fertile and poor sites separately. Furthermore, no trends were observed when plotting $\text{NE}_{\text{CO}_2 \text{ soil}}$ data from all sites, or from fertile and poor sites separately, against peat layer thickness or the stem volume, average age, and biomass increment of the tree stand.

3.2. Tree stand CO_2 sink

On the fertile sites, the tree stand CO_2 sink was on average $-880 \pm 60 \text{ g m}^{-2} \text{ year}^{-1}$ of CO_2 , and on the poor sites 45% smaller, $-490 \pm 60 \text{ g m}^{-2} \text{ year}^{-1}$ of CO_2 . The corresponding growth rates of stem volume were 6.9 and $3.9 \text{ m}^3 \text{ ha}^{-1} \text{ year}^{-1}$. The difference between the fertile and the poor sites was statistically significant (ANOVA, p , 0.00002) and the site type grouping explained 25% of the variation in sink. The tree stand CO_2 sink was positively correlated with temperature sum (Fig. 4; fertile: p , 0.01; r^2 , 0.19; poor: $p < 0.001$; r^2 , 0.50) and negatively with WT (Fig. 4; fertile: p , 0.003; r^2 , 0.25; poor: p , 0.035; r^2 , 0.14). Thus, the trends in the tree stand CO_2 sink were opposite to those in $\text{NE}_{\text{CO}_2 \text{ soil}}$.

3.3. Current greenhouse gas impact

Both the fertile and the poor sites had, on average, a climate cooling GHG impact (Fig. 5). On the fertile sites the sink was

Table 4

Mean above- and belowground litter production, decomposition and soil CO_2 balance ($\text{g m}^{-2} \text{ year}^{-1}$ of CO_2 , standard deviation in parentheses) for each site type (n in parenthesis). Site types: *Herb-rich* (Rhtkg), *Vaccinium myrtillus* I (Mtkg I) and II (Mtkg II), *Vaccinium vitis-idaea* I (Ptkg I) and II (Ptkg II), *Dwarf shrub* (Vatkg). Abbreviations are those Finnish abbreviations of Laine (1989). The first three site types fall under the group 'fertile' and the last three types under the group 'poor'.

Site type	Rhtkg (10)	Mtkg I (10)	Mtkg II (15)	Ptkg I (10)	Ptkg II (10)	Vatkg (13)
Aboveground litter	760 (270)	720 (200)	630 (260)	570 (210)	710 (260)	530 (190)
Belowground litter	800 (240)	820 (260)	710 (280)	820 (180)	690 (300)	550 (120)
Total litter	1560 (470)	1530 (350)	1340 (420)	1390 (350)	1410 (460)	1080 (270)
Decomposition	1740 (600)	1830 (470)	1470 (470)	1230 (360)	1350 (400)	1070 (310)
Soil CO_2 balance	180 (510)	300 (390)	130 (340)	–170 (120)	–50 (290)	–10 (130)

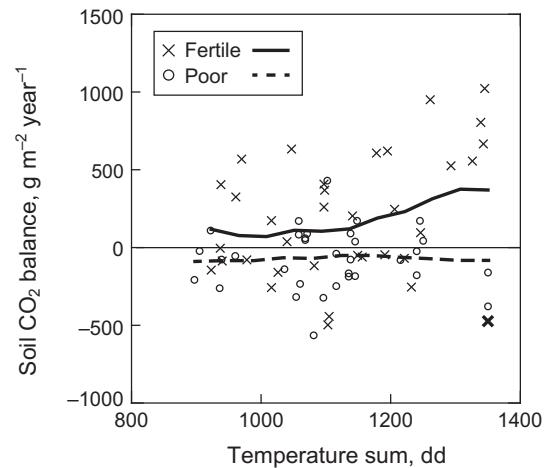


Fig. 2. Soil CO_2 balance versus temperature sum (threshold $+5^\circ\text{C}$). Sites are divided into two groups: fertile, including *Herb-rich* and *Vaccinium myrtillus* type sites, and poor, including *Vaccinium vitis-idaea* and *Dwarf shrub* type sites. Lines depict the running averages (window size = $n/2$). Positive value indicates source and negative value indicates sink. The fertile site marked in bold is unsuccessfully drained and almost in pristine condition.

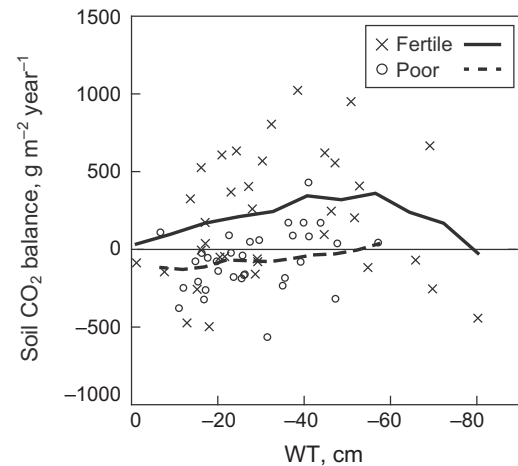


Fig. 3. Soil CO_2 balance versus mean water table (WT). Sites are divided into two groups: fertile, including *Herb-rich* and *Vaccinium myrtillus* type sites, and poor, including *Vaccinium vitis-idaea* and *Dwarf shrub* type sites. Lines depict the running averages (window size = $n/2$). Positive value indicates source and negative value indicates sink.

$-690 \pm 90 \text{ g m}^{-2} \text{ year}^{-1}$ of CO_2 eq. and on the poor sites somewhat lower, $-540 \pm 70 \text{ g m}^{-2} \text{ year}^{-1}$ of CO_2 eq. The soil GHG source at fertile sites ($+230 \pm 70 \text{ g m}^{-2} \text{ year}^{-1}$ of CO_2 eq.) compared to the soil GHG sink of the poor sites ($-50 \pm 40 \text{ g m}^{-2} \text{ year}^{-1}$ of CO_2 eq.) was overruled by the larger CO_2 sink of the tree stand. Both fertile and poor sites were small CH_4 and N_2O sources. The combined

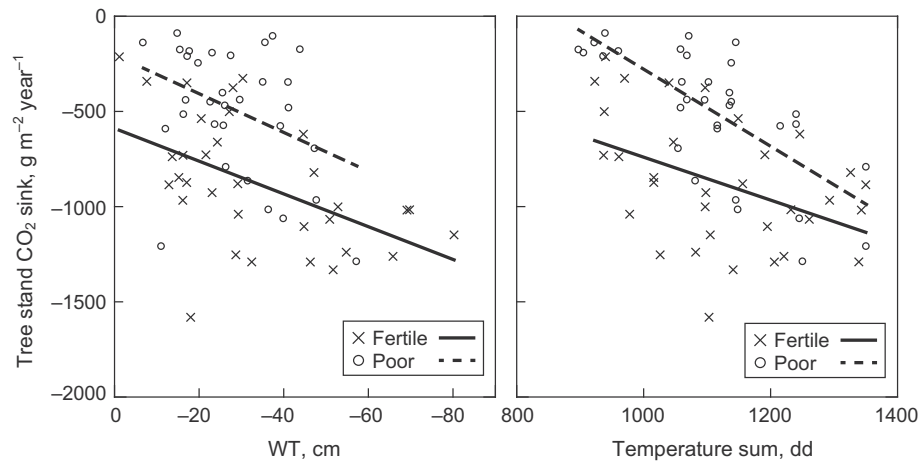


Fig. 4. Tree stand CO₂ sink versus mean water table (WT) and average annual temperature sum. Sites are divided into two groups: fertile, including *Herb-rich* and *Vaccinium myrtillus* type sites, and poor, including *Vaccinium vitis-idaea* and *Dwarf shrub* type sites. Lines depict linear regression.

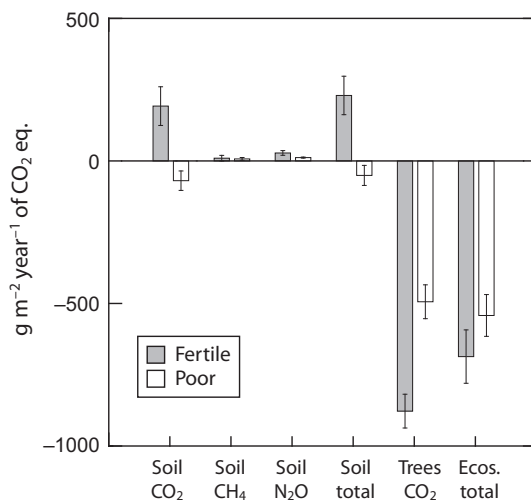


Fig. 5. Mean soil CO₂, CH₄ and N₂O and soil total greenhouse gas impact, tree stand CO₂ sink and total ecosystem greenhouse gas impact in CO₂ equivalents for fertile and poor sites. Error bars are \pm standard error of mean. Positive values are sources and negative values are sinks.

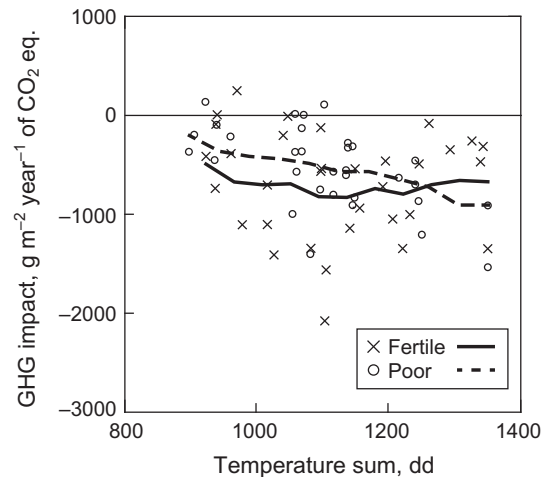


Fig. 6. Ecosystem greenhouse gas impact versus temperature sum. Sites are divided into two groups: fertile, including *Herb-rich* and *Vaccinium myrtillus* type sites, and poor, including *Vaccinium vitis-idaea* and *Dwarf shrub* type sites. Lines depict the running averages (window size = $n/2$). Positive value indicates source and negative value indicates sink.

source of those gases was $+40 \pm 10 \text{ g m}^{-2} \text{ year}^{-1}$ of CO₂ eq. on the fertile sites and $+20 \pm 5 \text{ g m}^{-2} \text{ year}^{-1}$ of CO₂ eq. on the poor sites.

Due to the contrary trends in the soil GHG source and the tree stand CO₂ sink, the ecosystem GHG balance of the fertile sites was independent of temperature sum (Fig. 6). On the poor sites, the GHG sink clearly increased with increasing temperature sum and exceeded that of the fertile sites in southernmost Finland. No correlation with WT was observed.

4. Discussion

4.1. Soil CO₂ balance

We found the main factors controlling soil CO₂ balance in boreal forestry-drained peatlands to be site fertility, water table, and temperature. WT is generally regarded as the main control of peatland CO₂ balance (e.g., Silvola et al., 1996; Couwenberg et al., 2011). The observation that while the fertile sites are CO₂ sources the poor sites may continue as CO₂ sinks even after drainage agrees with

earlier studies from forestry-drained peatlands (Minkkinen and Laine, 1998; Minkkinen et al., 1999; Lohila et al., 2011).

The difference in fertility can have several implications that contribute to the difference in the soil CO₂ balance: (1) Peat at the fertile sites may be easier to decompose than peat at the poor sites due to better nutrient availability. This has not been systematically tested, but for example Jaatinen et al. (2007) found higher basal respiration rates in the incubation of peat extracted from 0 to 10 cm depth at a mesotrophic than at an ombrotrophic forestry-drained peatland site. (2) The bulk density of peat is higher at fertile sites (mean 0.14 g cm^{-3}) than at poor ones (mean 0.10 g cm^{-3}). Thus, with the same WT, a higher mass of peat is available for aerobic decomposition at the fertile sites. (3) Ecological changes are more rapid and drastic in fertile sites (Laine et al., 1995; Minkkinen et al., 1999). Since less change occurs in the vegetation community of poor sites, the C accumulation function of a pristine peatland may continue; the moss layer remains, although some changes in species composition take place (Straková et al., 2010). Coverage of dwarf shrubs also remains high and may even increase (Laiho et al., 2003).

The climatic trend in our study was opposite to that of Minkkinen and Laine (1998). We found that the CO₂ source increased from north to south and that this trend was seen at fertile sites only, while they found an increasing C sink from north to south at all the studied site types. Both trends are plausible. *D* (soil respiration) is temperature dependent, thus higher temperatures lead to higher C output. On the other hand, litter production is also higher in the south (data not shown) leading to higher C input into soil. In addition to possible methodological uncertainties in the studies (Ojanen et al., 2012), a credible explanation to this discrepancy is that we studied the current CO₂ balance, whereas Minkkinen and Laine (1998) studied the cumulative change in C storage during several decades. As a result of drainage, ecological succession is still underway in the tree stand (Sarkkola et al., 2004, 2005) with consequent development of WT (Sarkkola et al., 2010) and gradual change in the species group composition and total biomass of ground vegetation (Laiho et al., 2003; Straková et al., 2012). Therefore, the long-term average C balance and current CO₂ balance do not necessarily equate.

The division between fertile and poor sites provides a means for the upscaling of soil CO₂ balance to larger areas. Further, the clear correlation with WT can be applied on poor sites. For fertile sites, this study does not give a definite answer as to how to distinguish between CO₂ source hot spots and more moderate sources. Rather, we have to content ourselves with the observation that in the south boreal vegetation zone sources of even 1000 g CO₂ m⁻² year⁻¹ are possible. More comprehensive data are needed for a more precise answer.

4.2. Current greenhouse gas impact

Soil CO₂ balance played a major role in the soil GHG impact. CH₄ and N₂O had only a minor effect, especially on the fertile sites. This is, however, true only at the site level. When considering larger areas, both fertile and poor sites co-exist. Then the CO₂ sink of poor sites and the CO₂ source of fertile sites will partly cancel each other out, and the relative importance of CH₄ and N₂O will become higher. If the proportions of poor and fertile peatlands are equal, as is approximately the case in Finland (Ylitalo, 2011), the share of CH₄ and N₂O in the soil GHG impact would be 30%. In addition, inclusion of CH₄ emissions from the drainage ditches (Minkkinen et al., 1997; Minkkinen and Laine, 2006) will further increase the importance of the non-CO₂ GHGs. Ditches with spacing of 40 m and width of 1 m cover 2.5% of the peatland area. Assuming that the ditches continue emitting CH₄ like pristine boreal peatlands, i.e., emissions ranging from 2 to 25 g CH₄ m⁻² year⁻¹ (e.g., Waddington and Roulet, 2000; Rinne et al., 2007; Leppälä et al., 2011), their combined effect on the CH₄ source is +1 to +16 g CO₂ eq. m⁻² year⁻¹. This would increase the share of CH₄ and N₂O in the GHG impact to 35%. Thus, CH₄ and N₂O should not be neglected when considering the GHG impact of forestry-drained peatlands.

Our results on soil GHG impact consider soils under a growing tree stand. After clearfelling, litter production of the tree stand ceases, but the decomposition in soil continues. It will take years before the litter production of the recovering ground vegetation and next tree generation will again turn the soil into a CO₂ sink (Mäkiranta et al., 2010). Also, logging residues may both enhance the decomposition of peat underneath and increase N₂O emissions (Mäkiranta et al., 2012). Site preparation, on the other hand, does not seem to have a considerable effect on the soil GHG balance (Pearson et al., 2012). All these effects need to be considered when estimating the long-term GHG impact of forestry drainage.

From the point of view of minimising the C loss from peat, the water table should be kept as close to the soil surface as possible. In Finnish forestry-drained peatlands, WT typically lies at –30 to –40 cm, which is also considered deep enough to sustain a well

growing tree stand (Sarkkola et al., 2010). With a WT of around –30 cm, the soil also turns from a CH₄ source into a CH₄ sink (Ojanen et al., 2010). Hence, this would be an optimal WT for a forestry-drained peatland, and excess drainage should be avoided.

Even though both the fertile and the poor sites were significant GHG sinks, the nature of the sink was distinctly dissimilar. Whereas at the poor sites both the soil and growing tree stand contributed to the sink, at the fertile sites the soil was a GHG source and the fast tree growth was needed to turn the ecosystem into a GHG sink. Thus, climatically sustainable ditching-based forestry on boreal peatlands seems to be possible on nutrient-poor sites. On fertile sites, the current high CO₂ sink of the growing tree stand can be considered as a means to mitigate climate change only if the harvested biomass is later stored, for example, in wooden buildings or as biochar in agricultural soils. The growth rates of stem volume (and consequently the tree stand CO₂ sinks) of both fertile and poor sites were of the same magnitude as in the 10th Finnish National Forest Inventory (Laine et al., 2012).

Rewetting of fertile peatlands could be a means to prevent soil CO₂ emissions. Yet, this does not necessarily mean that rewetting would provide a quick solution to mitigating climate change. Successful rewetting could restore the soil CO₂ sink, but at the same time the CO₂ sink of the growing tree stand is likely to reduce or even cease. Also, an increase in the CH₄ source is to be expected (Komulainen et al., 1998; Worrall et al., 2010). Furthermore, a peatland ecosystem functioning in a pristine manner will have a climate cooling effect only over millennia (Frolking et al., 2006; Frolking and Roulet, 2007). When planning the future use of forestry-drained peatlands, the net radiative forcing of different land use scenarios needs to be assessed.

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