

Long-term effect of fertilization on the greenhouse gas exchange of low-productive peatland forests



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ABSTRACT

Drainage of peatlands for forestry often leads to carbon dioxide (CO₂) net emission from soil due to loss of peat. This emission can be compensated for by the increased tree growth. However, many drained peatlands have low tree growth due to nutrient limitations. Tree growth at these peatlands can be effectively increased by fertilization, but fertilization has been also found to increase decomposition rates.

We studied the long-term effect of fertilization of low-productive forestry-drained peatlands on the complete ecosystem greenhouse gas exchange, including both soil and tree component, and accounting for CO₂, methane and nitrous oxide. Five N-rich study sites (flark fens and a rich fen) and one N-poor ombrotrophic site were established. Fertilization had started at the study sites 16–67 years before our measurements.

Fertilization considerably increased tree stand CO₂ sink (+248–1013 g CO₂ m⁻² year⁻¹). Decomposition increased on average by 45% (+431 g CO₂ m⁻² year⁻¹) and litter production by 38% (+360 g CO₂ m⁻² year⁻¹). Thus, on average 84% of the increased decomposition could be attributed to increased litter production and 16% to increased soil CO₂ net emission due to increased loss of peat. Soil CO₂ net emission correlated positively with water table depth and top soil N concentration.

Fertilization increased soil CO₂ net emission at the drained flark fens on average by 187 g CO₂ m⁻² year⁻¹. At the rich fen, net emission decreased. The N-poor bog exhibited soil CO₂ sink both with and without fertilization. Effects on methane and nitrous oxide emissions were small at most sites.

The increase in tree stand CO₂ sink was higher than the increase in soil CO₂ net emission, indicating that fertilization has a climate cooling effect in the decadal time scale. Yet, as the fertilized plots at N-rich sites exhibited soil CO₂ source or zero balance, continuation of fertilization-based forestry over several rotations would lead to progressive loss of ecosystem C. At the N-poor bog, fertilization-based forestry may have a climate-cooling effect also in the centennial time scale.

1. Introduction

Globally, approximately 15 million ha of peatlands have been drained for forestry (Paavilainen and Päivänen, 1995), which has influenced the capacity of these ecosystems to act as buffers against climate warming. In boreal peatlands, drainage for forestry (drainage hereafter) often leads to substantial and persistent loss of soil carbon (C) causing carbon dioxide (CO₂) net emissions to the atmosphere (Ojanen et al., 2013; Meyer et al., 2013; Minkkinen and Laine, 1998; Uri et al., 2017; Minkkinen et al., 2018). When drainage results in

increased tree growth, the CO₂ sink into the growing tree stand counterbalances the eventual soil CO₂ emissions until the tree stand is harvested (Hommeltnberg et al., 2014; Ojanen et al., 2013; Uri et al., 2017; Minkkinen et al., 2002).

Unfortunately, not all drainage has led to good forest growth. For example in Finland, 10–20% of the 5 million ha of drained peatlands is currently classified as low-productive and not profitable for commercial forestry use (Laiho et al., 2016). A substantial share of these peatlands are either nitrogen (N) rich, originally wet minerotrophic mires (0.2 mill. ha National Forest Inventory 11, Natural Resources Institute

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Finland) prone to nutrient deficiencies (for especially phosphorus P, potassium K, boron B) or extremely N-poor bogs (0.1 mill. ha) where also insufficient N availability severely hampers tree growth (Kaunisto, 1997).

The current greenhouse gas exchange of these N-rich and N-poor low-productive peatlands are poorly known, but estimates on their soil C loss during the first 30 years after drainage (Simola et al., 2012) indicate that substantial CO₂ net emissions from the soil are possible. While drainage increases soil respiration at such peatlands (Martikainen et al., 1995; Silvola et al., 1996), their productivity may even decrease (Vasander, 1982). Statistical models putting together the sporadic data indicate that heterotrophic soil respiration may indeed exceed litter production at the extremely N-poor bogs, leading to CO₂ net emissions from soil (Ojanen et al., 2014). For the N-rich low-productive peatlands, such estimates are not available. The rest of the low-productive peatlands (0.5 mill. ha) exhibit slow tree growth due a combination of cold climate and nutrient-poor soil and are unlikely to exhibit substantial CO₂ net emissions (Ojanen et al., 2013; Minkkinen and Laine, 1998).

Since tree growth in peatland ecosystems is generally limited by the lack of mineral nutrients, especially P and K (Holmen, 1964; Nieminen, 2000; Laiho et al., 2006), fertilizers are often added to increase productivity. For example in Finland, approximately 1.5 million ha of drained peatlands have been fertilized during the last decades (Saarsalmi and Mälkönen, 2001; Päivänen and Hännell, 2012; Finnish Statistical Yearbook of Forestry, 2013). At peatlands with high N availability, long-lasting increase in productivity has been observed following fertilization with PK fertilizers or wood ash (Moilanen et al., 2004). At bogs with low N availability, N also needs to be added to promote tree growth (Kaunisto, 1982; Moilanen and Issakainen, 1990).

While increasing the tree stand CO₂ sink, fertilization may also increase CO₂ emissions from the vast C storage of the peat soil, although no indications of increased CO₂ net emissions have been observed within the first five years after wood ash fertilization at boreal peatlands (Maljanen et al., 2006, 2014; Rütting et al., 2014; Klemedtsson et al., 2010; Ernfors et al., 2010). Long-term (8–55 years after fertilization) results show increased heterotrophic soil respiration *in situ* (Maljanen et al., 2014; Moilanen et al., 2012) and CO₂ production in laboratory incubation (Saarsalmi et al., 2014). Also, higher mass loss of cellulose samples, indicating accelerated decomposition rate have been observed (Moilanen et al., 2002, 2012; Saarsalmi et al., 2014). These long-term changes have been remarkable and consistent across the studied peatlands of varying fertility. Yet, the increased decomposition does not necessarily lead to increased CO₂ net emission from soil: fertilization increases net primary production (NPP) and most likely also litter input to soil. Increased heterotrophic soil respiration may indicate either increased decomposition of the increased litter pool or increased loss of peat. To estimate the net effect of fertilization on soil CO₂ exchange, changes both in litter production and heterotrophic soil respiration need to be measured.

Increased decomposition of peat could also increase nitrous oxide (N₂O) emissions as peat is generally N-rich. Yet, no increase in N₂O emissions has been observed in short or long term fertilization studies (Maljanen et al., 2006, 2014; Rütting et al., 2014; Klemedtsson et al., 2010; Ernfors et al., 2010). Data on long term effects of fertilization on N₂O emissions are, however, scarce (Maljanen et al., 2006, 2014). Fertilization has also potential to decrease methane (CH₄) emissions in the long term (Maljanen et al., 2006), as higher foliage biomass leads to increased canopy interception and transpiration and may provide sufficient biological drainage to prevent CH₄ emissions (Minkkinen et al., 2007; Sarkkola et al., 2010).

The aim of this study was to evaluate the potential of fertilization of N-rich and N-poor low-productive forestry-drained peatlands for climate change mitigation. They are promising targets for mitigation actions as they may currently have a climate warming effect due to soil CO₂ net emissions and slow tree growth, but fertilization can increase tree growth considerably. For this purpose, we estimated the long-term effect of fertilization on the ecosystem greenhouse gas (GHG) exchange at six peatland sites in south and middle boreal Finland. We asked the following questions: Are low-productive N-rich and N-poor drained peatland soils sources of CO₂ net emission? Does fertilization cause a long term increase in heterotrophic soil respiration at these peatlands and is this increase connected to an increase in litter production? How big are the GHG emissions of these peatlands and does fertilization increase or decrease emissions?

To our knowledge, this is the first study to estimate the long-term effect of peatland forest fertilization on the complete ecosystem GHG exchange, including both soil and tree stand, and accounting for all the GHGs relevant at peatland forests (CO₂, N₂O and CH₄).

2. Materials and methods

2.1. Study sites

The study material consisted of six study sites. They were low-productive drained peatlands, either N-rich or N-poor sites where long-term fertilization experiments with varying fertilizer types and doses had been carried out by the Natural Resources Institute Finland (the former Finnish Forest Research Institute) (Table 1). Prior to drainage, four of the sites were N-rich, minerotrophic flark fens (wet patterned fens) in Northern Finland. One N-rich site of rich fen origin was also included, as these peatlands form a distinct category on nutrient rich, calcareous bedrock areas in Northern Finland (Ruuhijärvi, 1983; Laitinen et al., 2017). In addition to N, rich fens are rich especially in calcium and magnesium and potentially highly productive, but the lack of P and K restricts tree growth (Nieminen and Penttilä, 2004; Silfverberg et al., 2011). The sixth site, N-poor *Sphagnum fuscum* pine bog, is a typical example of a low-productive drainage area in the raised bog region in Southern Finland. Differing from the other study sites, it has very low pH and is poor in all nutrients, including N (Kaunisto, 1982).

The study sites were selected from the population of fertilization experiments where fertilization had resulted in a well-growing forest stand. These experiments represent such fertilizations that are carried out also in practical forestry. At each site, one fertilized plot exhibiting typical tree growth for the treatment was selected as the fertilized plot (Fig. 1). The nearest unfertilized plot, within a few tens of meters, was selected as the control plot. Ditches adjacent to the control and fertilized plots were similarly and well maintained, except for the poor bog where ditches adjacent to the control plot had not been maintained and were thus partly blocked by vegetation. Based on soil N content and bulk densities (Fig. A1), the control and fertilized plot pairs had similar soil properties prior to fertilization.

Three of the sites had been treated with a single dose of wood ash and three of the sites had been repeatedly treated with PK and N fertilizers (Table 1). Time since the first fertilization varied from 16 to 67 years before our measurements. At Flark fen Ash16, that had received the lowest dose of nutrients (P 40 kg/ha, K 110 kg/ha), the fertilization dose was close to recommendations typical of productive peatland forests (P 45 kg/ha, K 80–90 kg/ha, Paavilainen, 1979). The other two ash-fertilized flark fens had received multifold doses of P and K compared to the recommendations. The sites fertilized with PK

Table 1
 Study sites and their treatment history. Naming: mire site type prior drainage (Laine et al., 2012), fertilization type, years since fertilization experiment started; id: identifier in the Natural Resources Institute Finland's field experiment database; coordinates ETRS89/WGS84. Drained peatland site types (Laine et al., 2012, see Päivänen and Hänel, 2012). Tree stand: stem volume (and increment), proportions of tree species of stem volume. Reference to published articles is included when available. Contr. = control plot, Fert. = fertilized plot.

Name	id	Coordinates	Original site type	Current site type	Peat depth (cm)		Tree stand stem volume (increment) m ³ /ha/(year)		Tree species (% of stem volume)		Drainage & operations	Fertilizations (kg/ha)
					Control	Fertilized	Contr.	Fert.	Contr.	Fert.		
Rich fen PK45 (Könölä)	9327	65° 57.77' N; 24° 29.6' E	Eutrophic birch fen (Kolk)	Herb-rich (Rhtkg II)	Herb-rich (Rhtkg II)	> 150	> 150	87 (4.3)	124 (5.6)	birch 66%, spruce 34%	birch 12%, spruce 88%	Drained 1939, amended 1981, ditch spacing 20–30 m, selective cutting of old birches 1979–1980 P 82 + K 239 [1969 PK 450 (P 41, K 77), 1984 KCI 170 (K 85), 1985 PK 450 (P 41, K 77)]
Flark fen Ash16 ^a (Rantaso)	9152	64° 47.325' N; 26° 29.951' E	Flark fen (RIN)	Cladina (Jätkg II)	Dwarf shrub (Vatkg II)	> 160	> 160	4 (0.1)	71 (3.7)	pine 100%	pine 100%	P 40 + K 110 [1998 Self-hardened wood ash 6500 (P 40, K 110)]
Flark fen Ash17 ^b (Pelso)	9144	64° 29.576' N; 26° 18.362' E	Herb-rich flark fen (RhRIN)	Dwarf shrub (Vatkg II)	Vacc. vitis-idaea (Ptkg II)	140	140	56 (1.6)	172 (9.4)	pine 100%	pine 100%	Drained 1955, amended 1994, ditch spacing 23–35 m, planted with pine 1960s P 135 + K 420 [1997 loose wood ash 15,000 (P 135, K 420)]
Flark fen PK47 (Jylkky)	3548, 3551	64° 52.969' N; 26° 06.068' E	Flark fen (RIN)	Dwarf shrub (Vatkg II)	Vacc. vitis-idaea (Ptkg II)	130	100	71 (2.3)	150 (7.0)	pine 97%, birch 3%	pine 62%, birch 32%, spruce 6%	Drained 1939, amended 1972, ditch spacing 20 m, thinned 1973 P 234 + K 218 [1967 PK 600 (P 102, K 90), 1973 PK 400 (P 96, K 60), 1984 PK 400 (P 96, K 68)]
Flark fen Ash67 ^c (Leppiniemi)	3726	64° 50.895' N; 26° 03.508' E	Herb-rich flark fen (RhRIN)	Dwarf shrub (Vatkg II)	Herb-rich (Rhtkg II)	> 160	100	37 (1.0)	203 (3.5)	pine 100%	pine 100%	Drained 1934, amended 2012, ditch spacing 30 m, Sown with pine 1934, thinned 1954, P 160–350 + K 530–1600 [1947 wood ash from birch logs 16,000 (P 160–350, K 530–1600)]
Poor bog NPK47 (Lyllynneva)	5633	62° 10.397' N; 22° 48, 627' E	<i>Sphagnum fuscum</i> bog (RaR)	Cladina (Jätkg I)	Dwarf-shrub (Vatkg I)	> 200	> 200	37 (1.2)	167 (5.8)	pine 100%	pine 100%	P 556 + K 435 + N 201 [1967 PK 2000 (P 340, K 300), 1975 PK 400 (P 96, K 60) & Ca-NH ₃ -NO ₂ 400 (N 110), 1991 (PK 500 (P 120, K 75) & Ca-NH ₃ -NO ₂ 330 (N 91))

^a Piirainen et al. (2013) (Utajärvi 1 & 3).

^b Moilanen et al. (2012).

^c Silfverberg and Hotanen (1989), Moilanen et al. (2002).

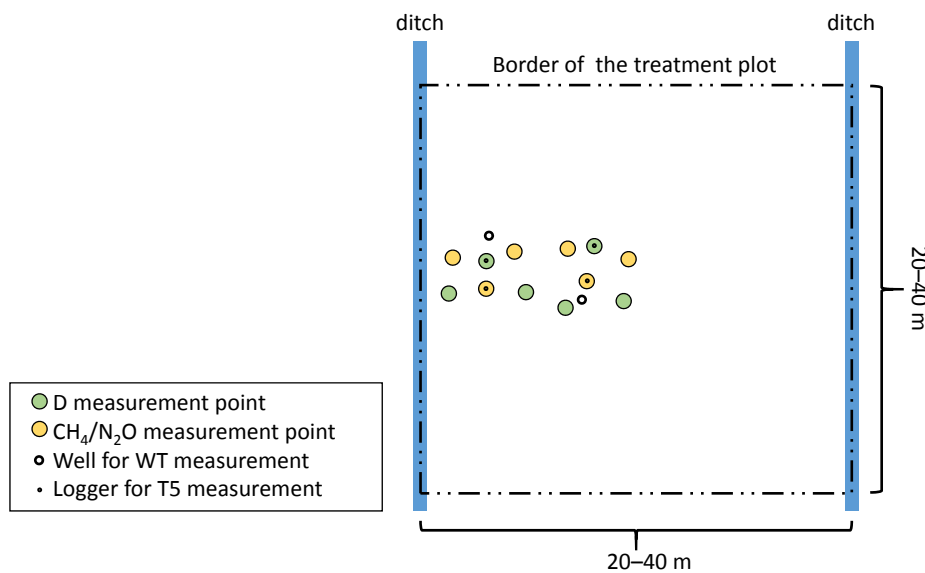


Fig. 1. Scheme for the systematic placement of the measurement points between ditch bank and mid-strip. Points were placed in the middle of the treatment plot, except at the fertilized and control treatment of Flark fen Ash16 and the control treatment of Poor bog NPK47, where the plot covered an area of 2–3 ha spanning over several strips. There, a location along a ditch within the treatment plot close to the other plot was selected to ensure that the measurements of both treatments were located as close to each other as possible.

fertilizers were given doses as recommended but repeated fertilizations had resulted in very high cumulative nutrient inputs. The two N fertilizations of the N-poor bog were at the recommended level of 100 kg/ha each (Paavilainen, 1979).

Wood ash, and from 1970s onwards also PK fertilizers, contain B. It is a micronutrient, the deficiency of which has often been found to restrict tree growth at drainage areas with thick peat layer and open fen origin (Veijalainen, 1983; Reinikainen and Veijalainen, 1983). Wood ash also contains a variety of other nutrients and it increases the pH of top soil and rate of N mineralization (Silfverberg and Hotanen, 1989; Moilanen et al., 2002).

The plots were classified to drained peatland site types according to Laine et al. (2012) based on ground vegetation composition (Table 1). The imbalance of nutrients had led to the development of the unfertilized control plots towards poorer drained site types than predicted by Laine et al. (2012). Thus, not only the tree stands but also ground vegetation exhibited signs of nutrient deficiency at the control plots. Parallel to the increase in the tree stand productivity, fertilization had resulted in a shift in the ground vegetation composition towards more fertile site types. Flark fen Ash67 showed the biggest change: the dwarf shrub and moss dominated vegetation at the control plot had been completely replaced by forb and graminoid dominated vegetation at the fertilized plot. The rich fen was different from the other N-rich sites: It exhibited the most fertile site type even at the control plot. The control plot of the poor bog fell expectedly under the least fertile site type.

2.2. Measurements and calculations

Six measurement points for heterotrophic soil respiration (decomposition D) and six points for CH₄ and N₂O fluxes were established at each plot in the spring of 2014 (Fig. 1). They were placed at even intervals between the ditch bank and the mid-strip or the next ditch to account for the likely systematic difference in water table between ditch bank and mid-strip. The measurement points were established by carving a 2-cm deep groove (d = 31.5 cm) in soil to enable the sealing of measurement chamber. For the D points, a galvanized steel cylinder (d = 40 cm, h = 50 cm) was inserted into soil to sever tree roots and to prevent their regrowth. Also living parts of mosses and aboveground

parts of vascular ground vegetation were removed to include only heterotrophic respiration. Possible regrowth of ground vegetation was removed before each measurement.

Two wells (perforated plastic pipes) were inserted into soil for water table depth (WT) measurements. WT was measured every time when gas fluxes were measured. Soil temperature was recorded (iButton DS1921G, Maxim Integrated, USA) at 5 cm depth (T5) at two decomposition and two CH₄/N₂O points at two-three hour intervals.

At the control plots of Flark fen PK47 (hummocks 80% of area) and Flark fen Ash67 (90%) with distinct hummock-lawn variation, half of the measurement points, temperature loggers and wells were installed at hummock and half at lawn surfaces. For those plots, annual fluxes were first calculated separately for hummocks and lawns. Then, the mean value of hummocks and lawns, weighted by their areal share was taken. Gas fluxes from ditches were not measured. As there were several plots with varying fertilization treatments along the same ditch at most sites, the difference in flux from the ditches adjacent to control and fertilized plots could not have been interpreted as an effect of the studied fertilization treatment.

Gas fluxes were measured altogether 11 times at each plot during the snow-free periods between July 2014 and September 2015. D was measured by a portable infrared gas analyzer (EGM-4, PP-Systems, USA) with an opaque closed chamber (modified SRC-1, d 31.5 cm, effective h 14.9 cm, PP-systems, USA; NSF11 in Pumpanen et al., 2004). Chamber closure time was 81 s and flux was calculated based on linear fit between CO₂ concentration and time. Gas samples for calculating CH₄ and N₂O fluxes were taken from an opaque closed chamber (d 31.5 cm, effective h 29.5 cm) that was equipped with a fan. Samples were taken from the chamber headspace at 5, 10, 15 and 20 min after placing the chamber at the point. Samples were analyzed in the laboratory of the Natural Resources Institute Finland Vantaa Unit with gas chromatograph (FI detector for CH₄ and EC detector for N₂O). Fluxes were calculated based on linear fit between gas concentration and time.

Annual fluxes were calculated separately for each point and plot-wise annual fluxes thereafter as means of all the points of the plot. For CH₄ and N₂O, fluxes for snow-free season (May–October) were calculated simply as mean of the measured fluxes, as there was no correlation

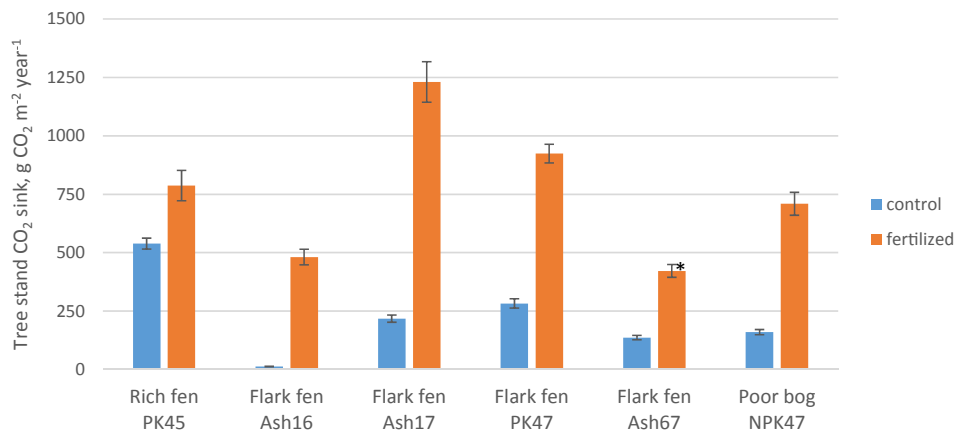


Fig. 2. Tree stand CO₂ sink (± standard error) at each control and fertilized plot. *The tree stand had been recently thinned very sparse the sink not representing the full production potential of the fertilized plot.

with T or WT. Snowy season (November–April) flux was calculated based on the relationship between mean snow-free and mean snowy season fluxes based on other studies (Appendix A). For D, annual fluxes for each point were calculated based on the relationship between measured D and T₅ at each point (modified from Lloyd and Taylor 1994):

$$D = D_{ref} e^{B \left(\frac{1}{T_{5ref} - T_{50}} - \frac{1}{T_5 - T_{50}} \right)} \quad (1)$$

where D_{ref} is a parameter describing D at the reference temperature T_{5ref} (here set at +10 °C) and B is a parameter describing the temperature sensitivity of D. T_{50} is the temperature at which D would reach 0 (here set at -15 °C).

Annual litter production (L) was calculated as the sum of aboveground L of tree stand, vascular ground vegetation and mosses and belowground L of roots and rhizomes. For all the components of L, C content of 50% was applied for the conversion from dry mass to CO₂. For the estimation of the aboveground L of tree stand, ten round litter collectors (d 28 cm) were placed on the ground at each plot along the transect. Litter collectors were emptied monthly during the snow-free season two years from spring 2014 to spring 2016. Litter was dried in 105 °C and weighed. Sum of the two-year litter mass was divided by two to estimate annual L.

For the estimation of the aboveground L of vascular ground vegetation, ten 50 × 50 cm² biomass samples were harvested during the maximum biomass in July–August 2016 at each plot. Biomass was separated by species and dried in 105 °C. L of herbaceous species was directly its biomass (renews annually). The biomass of dwarf shrub species was further separated to foliage and stems. L of dwarf shrub foliage was estimated as foliage mass divided by the number of annual whorls. L of dwarf shrub stems was estimated as stem mass divided by the mean age of stems. Vascular ground vegetation at the Rich fen PK45 could not be sampled. But as its role was minor (projection coverage 8.6% at the control plot and 6.0% at the fertilized plot), L was roughly estimated based on L/projection coverage ratios of herbaceous and arboreal ground vegetation from the other sites.

L of mosses was estimated as moss biomass production. Round nets (d 12.5 cm) were placed on colonies of the three most common moss species (three nets per species) at each site in the autumn 2014. Biomass grown through the nets was harvested one year later and dried in 105 °C. For *Sphagnum* species, masses and heights of capitula and

stems were estimated from a subsample. The harvested biomasses were corrected to account for the biomass harvested from the nets including capitula (that are not part of biomass production) instead of the corresponding length of stems (missing from biomass production). Plotwise L of mosses was calculated by multiplying the species specific biomass production estimates by respective projection coverages. For the species not sampled for biomass production at the respective plot, species-specific biomass production of the same species at another plot or a close species at the same plot was applied.

For the estimation of belowground L, 5–6 volumetric soil samples (area 15 × 15 cm² each, depth 0–20 cm) at each plot were cored. Roots and rhizomes were separated as follows: arboreal fine roots (trees and dwarf shrubs, d < 2 mm), coarse tree roots (d ≥ 2 mm), dwarf shrub and *Rubus chamaemorus* rhizomes + herb roots. Data was corrected to account for the 5.2% of arboreal fine roots and 7.4% of tree coarse roots found to be located in soil deeper than 20 cm in drained peatlands by Laiho and Finér (1996). L was calculated by multiplying biomasses with turnover ratios: arboreal fine roots 0.5 (Bhuiyan et al., 2017; Hansson et al., 2013; Leppälammii-Kujansuu et al., 2014), tree coarse roots 0.12 (Finér and Laine, 1998), dwarf shrub rhizomes 0.08 (Finér and Laine, 1998) and herb roots 1.25 (Laiho et al., 2003).

For CH₄ and N₂O, annual fluxes directly represent the net exchange, positive values meaning emission to the atmosphere. For CO₂, soil CO₂ net exchange was calculated based on annual D and L (Ojanen et al., 2012):

$$\text{Soil CO}_2 \text{ net exchange} = D - L, \quad (2)$$

positive value meaning net emission to the atmosphere. It should be kept in mind that these estimates are rather rough, as the measurements of D and L mainly cover only two years (instead of longer term average) and soil CO₂ net balance is sensitive to even relatively small changes in D and L (Ojanen et al. 2012). Note also that soil CO₂ net exchange does not equal soil C stock change, as waterborne C fluxes and CH₄-C flux are not included.

CO₂ sink due to tree stand biomass increment was estimated based on tree stand characteristics measured in the fall of 2015, except for Flark fen Ash17 and Flark fen Ash 67 where comparable measurements from 2010/2011 were available. Tree stand plots, 100–300 m² in area, covered the area where the GHG measurement points were located. Tree species and diameter (mm) at breast height (DBH) were recorded for all live trees (d ≥ 25 mm) and tree height, 5-year height increment, and

canopy length for sample trees (≥ 10 trees per plot) that were selected to represent the entire plotwise diameter distributions. For the sample trees, also DBH five years earlier (DBH_{-5}) were determined from increment cores. Those characteristics of the sample trees were regressed against their current DBH, and the obtained plotwise regressions were applied to estimate current tree height and crown length to all trees at each plot. Similarly, the respective tree-wise characteristics five years earlier were reconstructed by applying the regressions to the DBH_{-5} data. Species-specific single-tree biomass models of Repola (2008, 2009) and Laiho and Finér (1996) were applied to estimate current tree stand biomass and biomass five years earlier (similarly to Ojanen et al., 2012). Mean annual CO_2 sink to the biomass increment was estimated as the difference of the biomasses divided by 5.

Total plotwise GHG exchange was calculated by first converting CH_4 and N_2O exchange to CO_2 equivalents by multiplying with respective sustained global warming potentials (SGWP) and then subtracting the sum of equivalent soil GHG exchange from the tree stand CO_2 sink. Instead of the GWP values for pulse emissions suggested by IPCC (Myhre et al., 2013), we applied GWP values for sustained emissions (SGWP) as discussed by Neubauer and Megonigal (2015). SGWP values (47 for CH_4 , 270 for N_2O) for 100 year time horizon (proper time scale for forestry) were calculated, otherwise similarly to GWP values by IPCC but assuming that the emissions continue throughout the 100-year calculation period.

To roughly estimate the standard error of the plotwise GHG exchange, standard error of estimate was calculated for those components for which it was possible: For annual gas fluxes, it is the standard error of the mean of the pointwise annual fluxes. For moss L, it is based on the plotwise standard errors of the means of moss production of each species. For belowground L, it is the standard error of the mean of the sample specific Ls. For tree stand biomass, it includes parameter uncertainty and residual error of the biomass models (see Ojanen et al., 2012). These standard errors were aggregated (sum of uncorrelated

variances $var(a + b + \dots) = var(a) + var(b) + \dots$) to estimate the standard errors for GHG exchange.

The effect of fertilization on total GHG exchange was calculated for each site as:

$$GHG \text{ exchange of fertilized plot} - GHG \text{ exchange of control plot.} \quad (3)$$

3. Results

CO_2 sink of tree stand was remarkably higher at all fertilized plots compared to the control plots (Fig. 2). The smallest absolute ($+248 \text{ g } CO_2 \text{ m}^{-2} \text{ year}^{-1}$) and relative ($+46\%$) difference between fertilized and control plot was at the Rich fen PK45, where tree growth was high even at the control plot. At all the other sites, CO_2 sink of tree stand was at least three times as great at the fertilized plot, difference in sink ranging from $+285$ to $+1013 \text{ g } CO_2 \text{ m}^{-2} \text{ year}^{-1}$.

Both annual litter production (L) and decomposition (D) were consistently higher at the fertilized than at the control plots (Figs. 3 and 4). Rich fen was the only exception: D was lower at the fertilized plot. On average, L was 38% ($360 \text{ g } CO_2 \text{ m}^{-2} \text{ year}^{-1}$) higher and D was 45% ($431 \text{ g } CO_2 \text{ m}^{-2} \text{ year}^{-1}$) higher. Eighty four per cent of the mean increase in D could thus be attributed to increased litter input to soil, based on the assumption that long-term increase in litter input causes an equivalent increase in D.

The consistently higher aboveground litter production of the tree stands (on average $+254 \text{ g } CO_2 \text{ m}^{-2} \text{ year}^{-1}$, $+117\%$) was the main reason for the higher L at the fertilized plots (Fig. 3). On the other hand, the belowground arboreal litter production was very variable, and fertilized plots showed both larger and smaller values compared to control plots. Litter production of mosses and aboveground parts of vascular ground vegetation was smaller at fertilized compared to control plots at most sites. The difference in both above and below ground litter production of herbaceous litter was exceptional at Flark fen

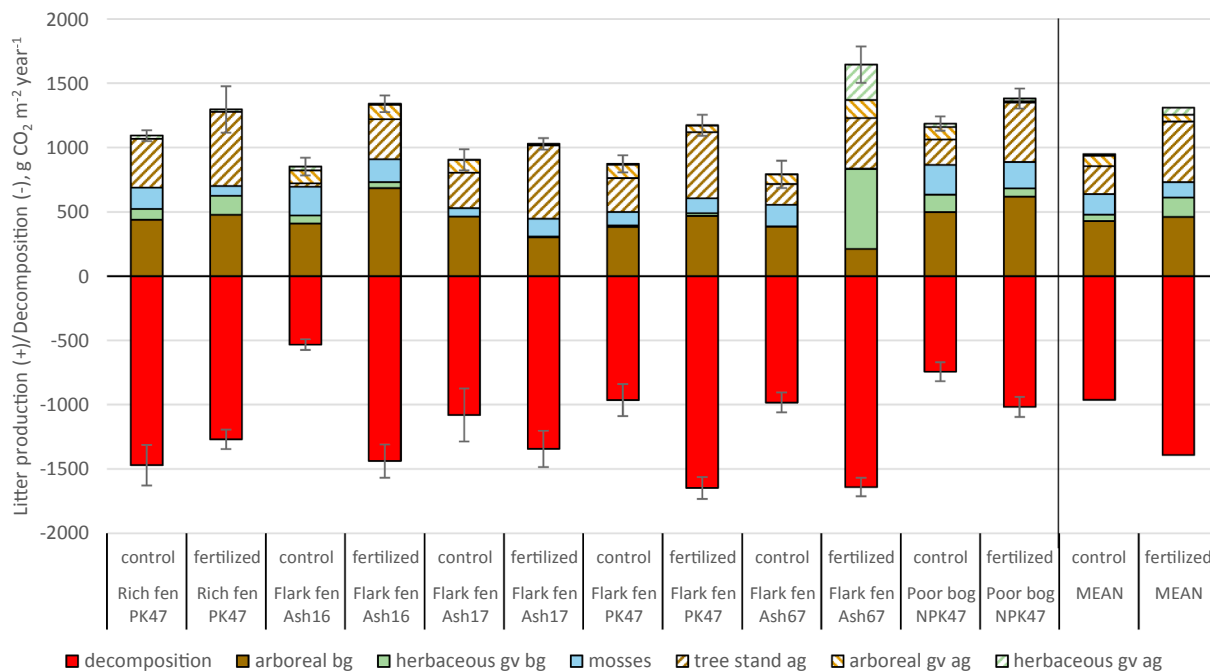


Fig. 3. Decomposition and the components of litter production at each control and fertilized plot. ag = aboveground, bg = belowground, gv = ground vegetation. Error bars are \pm standard error of L and D.

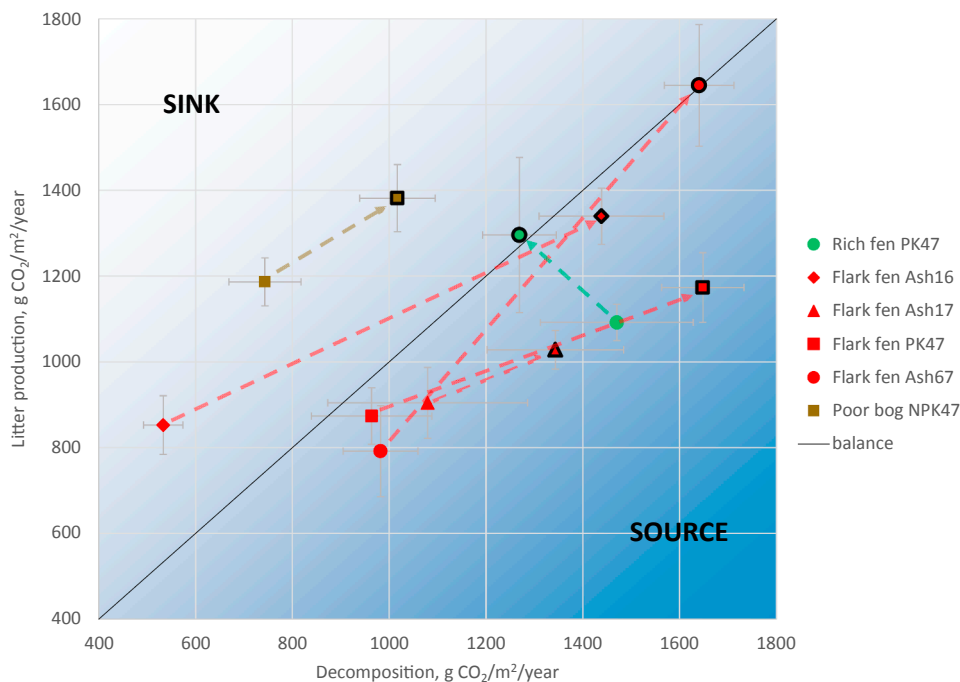


Fig. 4. Litter production versus decomposition (± standard error) at the control (no border)–fertilized (black border) plot pairs.

Ash67, where dwarf shrubs dominated at the control plot but herbaceous species at the fertilized plot.

D was clearly higher at fertilized plots than at control plots on three of the flark fen sites while minor differences were found on the rest of the sites (Fig. 4). Of these sites, only at the Flark fen Ash67 this difference was accompanied by a corresponding difference in litter production, resulting in no change in soil CO₂ exchange. This is also the only site where fertilization had changed site characteristics completely, as revealed by both a very large increase in soil bulk density (Fig. A2) and a drastic contrast in ground vegetation with dwarf shrub site type on the control plot and herb-rich type on the fertilized plot

(Figs. 3 and 5).

Soil was CO₂ neutral or source to atmosphere at most of the plots (Figs. 4 and 5). The highest emission, 475 ± 118 g CO₂ m⁻² year⁻¹, occurred at the fertilized plot of Flark fen PK47. Only the control plot of Flark fen Ash16 and both plots of Poor bog NPK47 showed a clear soil CO₂ sink. On average, fertilization increased soil CO₂ emissions: the average soil CO₂ emission of the flark fens was 34 g CO₂ m⁻² year⁻¹ at the control plots and 221 g CO₂ m⁻² year⁻¹ at the fertilized plots. Soil CO₂ emission correlated positively with WT and top soil N concentration (Fig. 5).

CH₄ and N₂O emissions and their responses to fertilization were

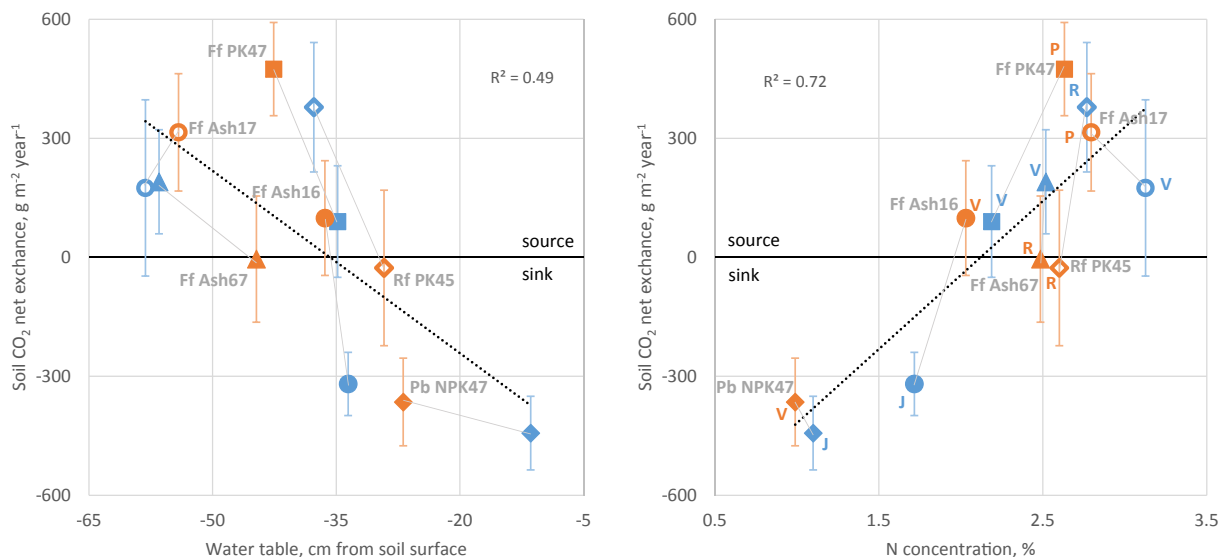


Fig. 5. Soil CO₂ net exchange (± standard error) versus mean water table depth (WT) and nitrogen concentration of the top 20 cm of soil at the control (blue) and fertilized (orange) plots. Positive values indicate CO₂ emission to the atmosphere. Different symbols indicate the study sites: Ff = flark fen, Rf = rich fen, Pb = poor bog. Letters denote the drained peatland site types (Laine et al., 2012, from the least to most fertile: J Cladina type (Jätkg), V Dwarf shrub type (Vatkg), P *Vaccinium vitis-idaea* type (Ptkg), M *Vaccinium myrtillus* type (Mtkg), R Herb-rich type (Rhtkg)).

negligible except for two sites (Fig. 5). At the poor bog, CH₄ emission was high ($1.7 \pm 1.1 \text{ g CH}_4 \text{ m}^{-2} \text{ year}^{-1}$) at the control plot, apparently due to the exceptionally high WT caused by the partially blocked ditches. At the Flark fen Ash17, N₂O emission at the control plot was exceptionally high ($1.14 \pm 0.36 \text{ g N}_2\text{O m}^{-2} \text{ year}^{-1}$), coinciding with the very high N concentration in the top soil. There, emission at the fertilized plot ($0.21 \pm 0.03 \text{ g N}_2\text{O m}^{-2} \text{ year}^{-1}$) was much smaller, even though markedly higher than at the other fertilized plots.

Both GHG sources and sinks occurred at control plots (Fig. 7A). Conversely, all the fertilized plots were GHG sinks, as the clearly increased tree stand CO₂ sink overruled the eventual soil CO₂ net emission (Fig. 7B). Poor bog NPK47 had the highest GHG sink both among control and fertilized plots, as both tree stand and soil had a substantial contribution to the sink. Fertilization decreased GHG emissions at all the sites (Fig. 7C). At the Flark fen Ash16 the decrease was minor, as the substantial increase in soil CO₂ emission almost counterbalanced the increased tree stand CO₂ sink. On average, the difference in GHG exchange between control and fertilized plots was $-527 \text{ g CO}_2 \text{ eq. m}^{-2} \text{ year}^{-1}$.

4. Discussion

Fertilization of low-productive peatlands has potential for climate change mitigation in the decadal time scale. This study revealed that the great increase in productivity due to fertilization leads to a long-term increase in tree stand CO₂ sink that clearly exceeds the increase in soil CO₂ net emissions (Figs. 7 and 8). Also, no short-term increase in soil GHG emissions at forestry-drained peatlands have been observed in earlier studies (Maljanen et al., 2006, 2014; Rütting et al., 2014; Klemetsson et al., 2010; Ernfors et al., 2010).

Even though promising in the decadal time scale, fertilization of low-productive peatlands likely has harmful climate impacts in the centennial time scale. Unfortunately, when well-drained and fertilized, the studied N-rich peatlands were a net source of CO₂ from soil (Fig. 8), similarly to other well-drained highly productive peatland forests (Meyer et al., 2013; Ojanen et al., 2013; Uri et al., 2017). This indicates loss of peat C to the atmosphere and undermines the rationale of peatland fertilization as a means to mitigate climate change. Continued soil CO₂ net emission and eventual cuttings of tree stand would unavoidably lead to progressive loss of ecosystem C to the atmosphere. For the soil C storage to be sustained over complete forest rotations, soil under growing tree stand should be a CO₂ sink, as there is a period of substantial soil C loss after final felling when litter input to soil drops but decomposition continues (Mäkiranta et al., 2010; Pearson et al.,

2012). Further, for a full assessment of the effect of fertilization on climate, also greenhouse gas emissions from the production and transportation of fertilizers should be included.

Rewetting by blocking the ditches could be a feasible future land-use option for the N-rich peatlands. This would undoubtedly lead to long-term protection of the peat C store (Wilson et al., 2016) and eventually to peat accumulation (Kareksela et al., 2015). Yet, as rewetting would result in increased CH₄ emissions (Wilson et al., 2016; Koskinen et al., 2016) and in at least partial cessation of the tree stand CO₂ sink, it can result even in a climate warming effect in the decadal time scale. For example, the CH₄ emission of a pristinely functioning peatland offsets the CO₂ sink to peat accumulation to the extent that achieving a climate cooling impact (negative radiative forcing) may take centuries to millennia (Frolking et al., 2006).

The extremely N-poor drained bog told a different story. In accordance to findings on productive nutrient poor peatland forests (Ojanen et al., 2013; Minkkinen et al., 2018), the soil was a CO₂ sink. Heavy fertilization (Table 1) with the associated deeper WT did not compromise this sink: the very low soil N content and pH likely still restricted decomposition. Thus, fertilization of such sites may have potential for even long-term climate change mitigation. If fertilization is not considered economically feasible, these peatlands could be abandoned from forestry and eventually let to rewet spontaneously.

For the N-rich flark fens, this study corroborated the findings of earlier studies on the long-term effects of fertilization: decomposition and tree stand CO₂ sink increased, while changes in CH₄ and N₂O emissions were generally small (Maljanen et al., 2006, 2014; Moilanen et al. 2002, 2012; Saarsalmi et al., 2014). Even though a major part of the increased soil respiration was compensated by the increase in litter production, fertilization on average increased soil net CO₂ emission. Owing to the substantially increased tree stand CO₂ sink, ecosystem GHG emissions decreased or were unaltered.

Comparison to a range of productive drainage areas in Northern Finland (Ojanen et al., 2013; Fig. 8) shows that fertilization may convert the low-productive flark fens to highly productive peatland forests. Also decomposition shifted from the level of poorly-drained to that of well-drained peatland forests. Soil at the fertilized plots was a source of CO₂ of the same magnitude than at other well-drained productive sites, while poorly-drained productive forests often exhibit soil CO₂ sink (Fig. 8). At the ecosystem level, the fertilized plots of the present study were among the biggest CO₂ sinks while the control plots were among the smallest ones.

The rich fen behaved differently: unlike in the N-rich flark fens, fertilization did not increase decomposition. This seemed to be due to

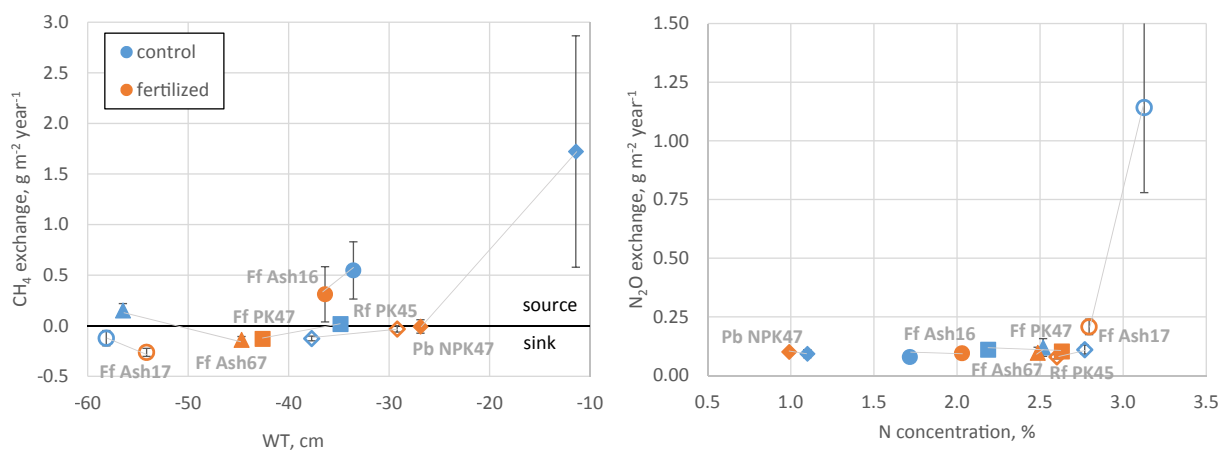


Fig. 6. Soil CH₄ exchange (\pm standard error) versus mean water table level (WT) and N₂O exchange versus nitrogen concentration of the top 20 cm of soil at the control (blue) and fertilized (orange) plots. Positive values indicate CH₄/N₂O emission to the atmosphere. Different symbols indicate the study sites: Ff = flark fen, Rf = rich fen, Pb = poor bog.



Fig. 7. Greenhouse gas exchange (\pm standard error) at drained (A) and fertilized (B) plots and the effect of fertilization (C) as CO₂ equivalents at the six sites. Positive values indicate GHG emission to the atmosphere (A, B) or increase in emission (C).

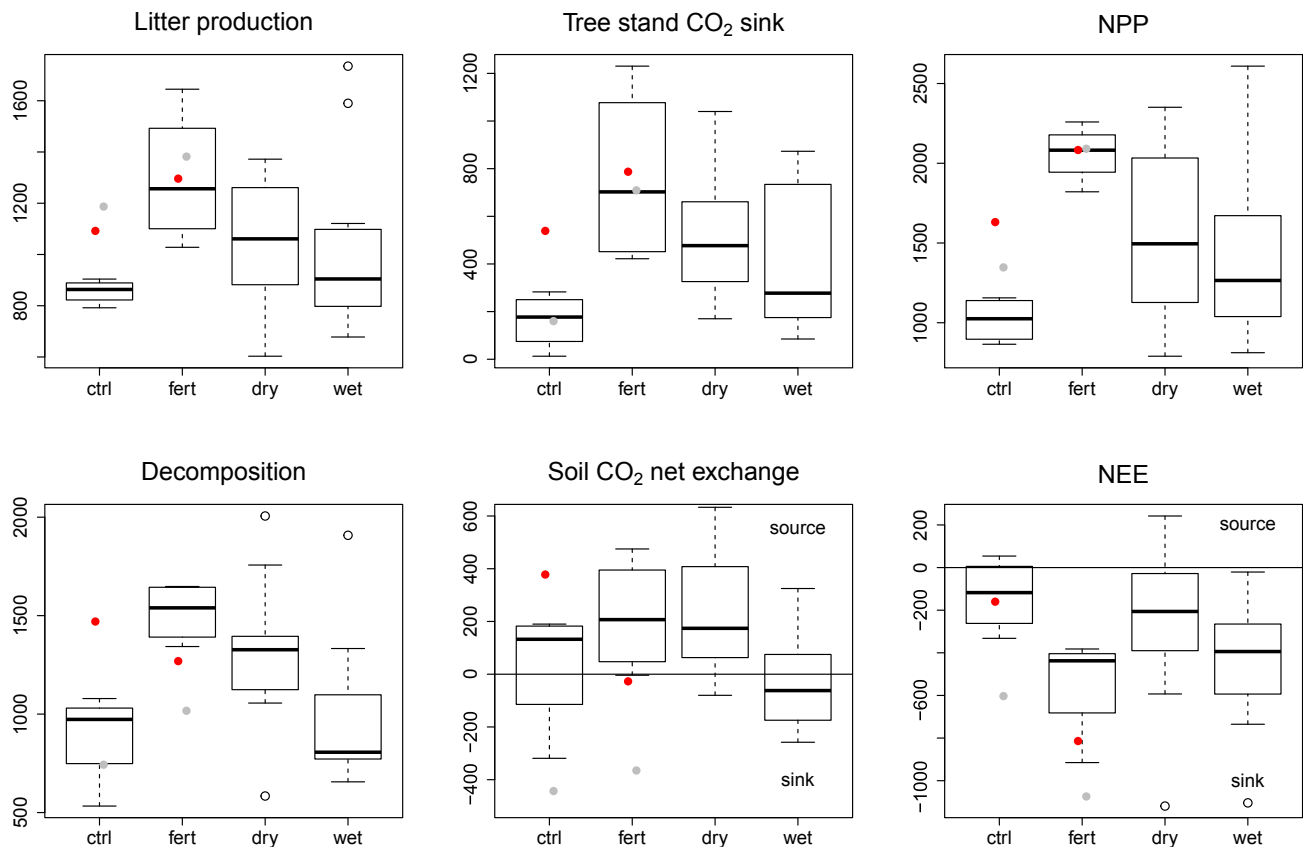


Fig. 8. Comparison of litter production, tree stand CO₂ sink, net primary production (NPP = litter production + tree stand CO₂ sink), Decomposition, soil CO₂ net exchange (= Decomposition – litter production) and net ecosystem exchange (NEE = soil CO₂ net exchange – tree stand CO₂ sink) between productive drainage areas in Northern Finland (Ojanen et al., 2013) grouped into well-drained (“dry”, WT range –23 to –53 cm, n = 9) and poorly drained (“wet”, WT range –18 to –1 cm, n = 12) sites and the control (ctrl) and fertilized (fert) plots of the flark fens (n = 4), the rich fen (red dot) and the poor bog (grey dot) of this study. Unit = g CO₂ m⁻² year⁻¹.

productivity as well as decomposition being at a relatively high level even at the control plot, both compared to our other N-rich study sites and earlier observations from productive drained peatlands (Fig. 8). Clearly, the decomposition of peat was not severely restricted by the site’s nutrient regime even without fertilization.

The effect of fertilization on CH₄ emissions was generally negligible (Fig. 7C), but it must be noted that the ditches at our study sites had been exceptionally well maintained. Generally, ditch maintenance at low-productive drainage areas has not been carried out since their initial drainage, which occurred decades ago at most of the drained peatlands in Finland (Päivänen and Hännel 2012). Thus, a small CH₄ emission reduction to the extent observed for the poor bog in this study would be expected due to improved drainage (Ojanen et al., 2010). CH₄ emissions from ditches would also be reduced if ditches were cleaned in addition to fertilization (Minkkinen et al., 1997; Minkkinen and Laine, 2006).

Confirming earlier findings by Maljanen et al. (2006, 2014), fertilization does not seem to induce a risk of N₂O emissions. Even though several of the control and fertilized N-rich plots of this study showed considerable soil CO₂ net emissions indicating loss of peat and suggesting N mineralization, substantial N₂O emission occurred only at one site (Fig. 6). This is in line with earlier observations showing that emissions of up to 1 g N₂O m⁻² year⁻¹ occur sporadically at N-rich

sites, while on the majority of forestry-drained peatlands emissions are very low (Ojanen et al., 2010, 2018). The greatly smaller N₂O emissions at the fertilized plot of the N-richest sites suggest that the emissions may even decrease if they have been high. This can be because the N demand of the vegetation must be much higher at the fertilized plot compared to the control plot: NPP was twice as high at the fertilized plot compared to the control plot. Thus, while fertilization may increase N mineralization (Moilanen et al., 2002) through enhanced decomposition, also NPP increases leading to increased N demand.

This study is an example of the global trend that turning peatlands into highly productive agricultural and forestry systems by intensive drainage and fertilization often leads to gradual loss of peat C storage (e.g. Wilson et al., 2016). Over centuries, vast amounts of CO₂ may be released to the atmosphere. But this study also underlines that decadal climate change mitigation goals and longer-term conservation of peat C storage may compromise each other at Northern forestry-drained peatlands. As the annual soil CO₂ net emissions at these peatlands are much smaller than under more intensive land-uses and warmer climates (see, e.g. Wilson et al., 2016), CO₂ sink into the growing tree stand may more than compensate for soil emissions for decades. Careful consideration is needed when giving recommendations for the future land-use of these drained peatlands.

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Appendix A

See Figs. A1 and A2.

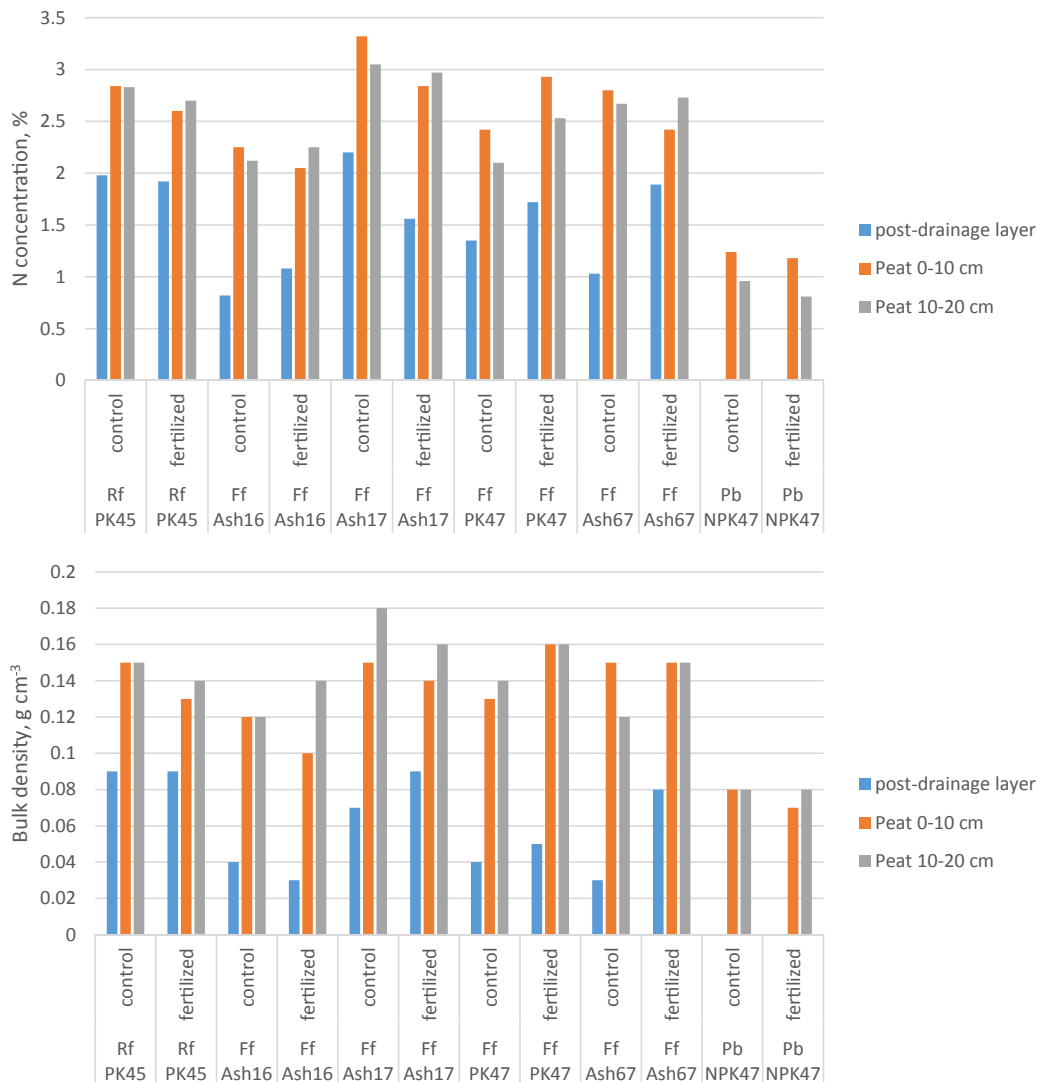


Fig. A1. Soil nitrogen (N) concentration and bulk density at the control and fertilized plot of each study sites. Post-drainage layer is the layer of organic soil accumulated on top of the pre-drainage peat. The two peat layers are 0–10 cm and 10–20 cm below the bottom of the post-drainage layer. At Poor bog NPK47, a separate post-drainage layer could not be distinguished.

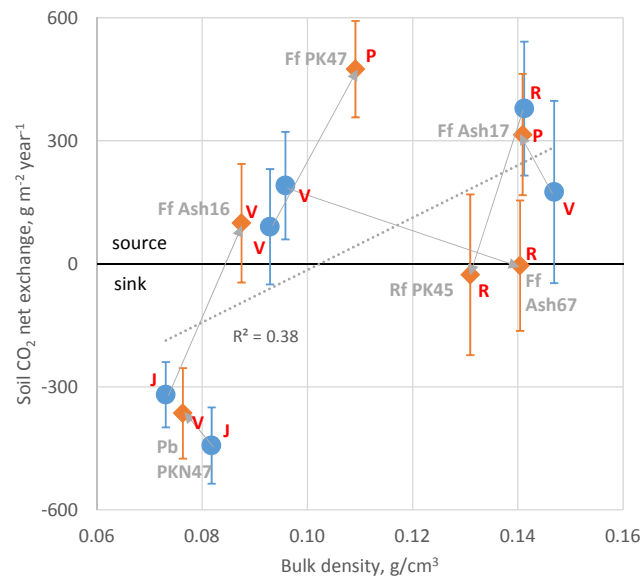


Fig. A2. Soil CO₂ net exchange (± standard error) versus bulk density in top 20 cm of soil at the control (blue) and fertilized (orange) plots. Positive values indicate CO₂ emission to the atmosphere. Ff = flark fen, Rf = rich fen, Pb = poor bog. The red letters denote the drained peatland site types (Laine et al., 2012, from the least to most fertile: J Cladina type (Jätkg), V Dwarf shrub type (Vatkg), P *Vaccinium vitis-idaea* type (Ptkg), M *Vaccinium myrtillus* type (Mtkg), R Herb-rich type (Rhtkg)).

Appendix B. Supplementary material

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.trd.2018.09.020>.

References

- Bhuiyan, R., Minkkinen, K., Helmisaari, H.-S., Ojanen, P., Penttilä, T., Laiho, R., 2017. Estimating fine-root production by tree species and understorey functional groups in two contrasting peatland forests. *Plant Soil* 412, 299–316. <https://doi.org/10.1007/s11104-016-3070-3>.
- Ernfors, M., Sikström, U., Nilsson, M., Klemedtsson, L., 2010. Effects of wood ash fertilization on forest floor greenhouse gas emissions and tree growth in nutrient poor drained peatland forests. *Sci. Tot. Environ.* 408, 4580–4590. <https://doi.org/10.1016/j.scitotenv.2010.06.024>.
- Finér, L., Laine, J., 1998. Root dynamics at drained peatland sites of different fertility in southern Finland. *Plant Soil* 201, 27–36.
- Finnish Statistical Yearbook of Forestry, 2013. Finnish Forest Research Institute. pp. 449.
- Frolking, S., Roulet, N., Fuglestedt, J., 2006. How northern peatlands influence the Earth's radiative budget: Sustained methane emission versus sustained carbon sequestration. *J. Geophys. Res.* 111, G01008. <https://doi.org/10.1029/2005JG000091>.
- Hansson, K., Helmisaari, H.-S., Sah, S., Lange, H., 2013. Fine root production and turnover of tree and understorey vegetation in Scots pine, silver birch and Norway spruce stands in SW Sweden. *For. Ecol. Manage.* 309, 58–65. <https://doi.org/10.1016/j.foreco.2013.01.022>.
- Holmen, H., 1964. Forest ecological studies on drained peat land in the province of Uppland, Sweden. *Parts I-III. Stud. Forestalia Suecica* 16, 1–235.
- Hommelntenberg, J., Schmid, H., Drösler, M., Werle, P., 2014. Can a bog drained for forestry be a stronger carbon sink than a natural bog forest? *Biogeosci.* 11, 3477–3493. <https://doi.org/10.5194/bg-11-3477-2014>.
- Kareksela, S., Haapalehto, T., Juutinen, R., Matilainen, R., Tahvanainen, T., Kotiaho, J., 2015. Fighting carbon loss of degraded peatlands by jump-starting ecosystem functioning with ecological restoration. *Sci. Tot. Environ.* 537, 268–276. <https://doi.org/10.1016/j.scitotenv.2015.07.094>.
- Kaunisto, S., 1982. Development of pine plantations on drained bogs as affected by some peat properties, fertilization, soil preparation and liming. Available at: *Comm. Inst. For. Fenn.* 109, 1–56. <http://urn.fi/URN:ISBN:951-40-0597-X>.
- Kaunisto, S., 1997. Peatland forestry in Finland: problems and possibilities from the nutritional point of view. In: Trettin, C.C., Jurgensen, M.F., Grigal, D.F., Gale, M.R., Jeglum, J.K. (Eds.), *Northern forested Wetlands: Ecology and Management*. CRC Press, Boca Raton, pp. 387–401.
- Klemedtsson, L., Ernfors, M., Björk, R., Weslien, P., Rütting, T., Grill, P., Sikström, U., 2010. Reduction of greenhouse gas emissions by wood ash application to a Picea abies (L.) Karst. forest on a drained organic soil. *Eur. J. Soil Sci.* 61, 734–744. <https://doi.org/10.1111/j.1365-2389.2010.01279.x>.
- Koskinen, M., Maanavilja, L., Nieminen, M., Minkkinen, K., Tuittila, E.-S., 2016. High methane emissions from restore Norway spruce swamps in southern Finland over one growing season. *Mire Peat* 17 (2), 1–13. <https://doi.org/10.19189/Map.2015.OMB.202>.
- Laiho, R., Finér, L., 1996. Changes in root biomass after water-level drawdown on pine mires in southern Finland. *Scand. J. For. Res.* 11, 251–260.
- Laiho, R., Penttilä, T., Vasander, H., Laine, J., Alm, J., 2006. Post-drainage dynamics of P uptake by trees and ground vegetation in Scots pine dominated peatlands. *Suo* 57, 1–10.
- Laiho, R., Tuominen, S., Kojola, S., Penttilä, T., Saarinen, M., Ihalainen, A., 2016. Heikkotuottoiset ojitetut suometsät – missä ja paljonko niitä on? *Metsätieteen Aikakauskirja* 2 (2016), 73–93.
- Laiho, R., Vasander, H., Penttilä, T., Laine, J., 2003. Dynamics of plant-mediated organic matter and nutrient cycling following water-level drawdown in boreal peatlands. *Glob. Biogeochem. Cycles* 17, 1053.
- Laitinen, J., Oksanen, J., Kaakinen, E., Parviainen, M., Küttim, M., Ruuhijärvi, R., 2017. Regional and vegetation-ecological in northern boreal flark fens of Finnish Lapland: analysis from a classic material. *Ann. Bot. Fenn.* 54, 179–195. <https://doi.org/10.5735/085.054.0327>.
- Laine, J., Vasander, H., Hotanen, J.-P., Nousiainen, H., Saarinen, M., Penttilä, T., 2012. Suotyypit ja Turvekankaat – opas Kasvupaikkojen Tunnistamiseen. *Metsäkustannus Oy, Helsinki*, pp. 160.
- Leppälampi-Kujansuu, J., Salemaa, M., Berggren Kelja, D., Linder, S., Helmisaari, H.-S., 2014. Fine root turnover and litter production of Norway spruce in a long-term temperature and nutrient manipulation experiment. *Plant Soil* 374, 73–88. <https://doi.org/10.1007/s11104-013-1853-3>.
- Lloyd, J., Taylor, J.A., 1994. On the temperature dependence of soil respiration. *Funct. Ecol.* 8, 315–323.
- Maljanen, M., Jokinen, H., Saari, A., Strömmer, R., Martikainen, P.J., 2006. Methane and nitrous oxide fluxes, and carbon dioxide production in boreal forest soil fertilized with wood ash and nitrogen. *Soil Use Manage.* 22, 151–157. <https://doi.org/10.1111/j.1475-2743.2006.00029.x>.
- Maljanen, M., Liimatainen, M., Hytönen, J., Martikainen, P.J., 2014. The effect of granulated wood-ash fertilization on soil properties and greenhouse gas (GHG) emissions in boreal peatland forests. *Bor. Environ. Res.* 19, 295–309.
- Mäkiranta, P., Riutta, T., Penttilä, T., Minkkinen, K., 2010. Dynamics of net ecosystem CO₂ exchange and heterotrophic soil respiration following clearfelling in a drained peatland forest. *Agric. For. Met.* 150, 1585–1596. <https://doi.org/10.1016/j.agrformet.2010.08.010>.
- Martikainen, P.J., Nykänen, H., Alm, J., Silvola, J., 1995. Change in fluxes of carbon dioxide, methane and nitrous oxide due to forest drainage of mire sites of different trophy. *Plant Soil* 168–169, 571–577.
- Meyer, A., Tarvainen, A., Nouratpour, A., Björk, R., Ernfors, M., Grelle, A., Kasimir Klemedtsson, Å., Lindroth, A., Rantfors, M., Rütting, T., Wallin, G., Weslien, P., Klemedtsson, L., 2013. A fertile peatland forest does not constitute a major greenhouse gas sink. *Biogeosci.* 10, 7739–7758. <https://doi.org/10.5194/bg-10-7739-2013>.
- Minkkinen, K., Korhonen, R., Savolainen, J., Laine, J., 2002. Carbon balance and radiative forcing of Finnish peatlands 1990–2100 – the impact of forestry drainage. *Glob. Change Biol.* 8, 785–799.
- Minkkinen, K., Laine, J., 1998. Long-term effect of forest drainage on the peat carbon stores of pine mires in Finland. *Can. J. For. Res.* 28, 1267–1275.
- Minkkinen, K., Laine, J., 2006. Vegetation heterogeneity and ditches create spatial variability in methane fluxes from peatlands drained for forestry. *Plant Soil* 285, 289–304. <https://doi.org/10.1007/s11104-006-9016-4>.
- Minkkinen, K., Laine, J., Hykänen, H., Martikainen, P.J., 1997. Importance of drainage ditches in emissions of methane from mires drained for forestry. *Can. J. For. Res.* 27,

- 949–952.
- Minkkinen, K., Ojanen, P., Penttilä, T., Aurela, M., Laurila, T., Tuovinen, J.-P., Lohila, A., 2018. Persistent carbon sink at a boreal drained bog forest. *Biogeosci.* 15, 3603–3624. <https://doi.org/10.5194/bg-15-3603-2018>.
- Minkkinen, K., Penttilä, T., Laine, J., 2007. Tree stand volume as a scalar for methane fluxes in forestry-drained peatlands. *Bor. Environ. Res.* 12 (2), 127–132.
- Moilanen, M., Hytönen, J., Leppälä, M., 2012. Application of wood ash accelerates soil respiration and tree growth on drained peatland. *Eur. J. Soil Sci.* 63, 467–475. <https://doi.org/10.1111/j.1365-2389.2012.01467.x>.
- Moilanen, M., Issakainen, J., 1990. Suometien PK-lannos ja typpilannoitelajit karuhkojen ojitettujen rämeiden lannoituksessa. (Summary: PK fertilizer and different types of N fertilizer in the fertilization of infertile drained pine bogs. Available at: *Folia Forestalia* 754, 1–20. <http://urn.fi/URN:ISBN:951-40-1112-0>).
- Moilanen, M., Silfverberg, K., Hökkä, H., Issakainen, J., 2004. Comparing effects of wood ash and commercial PK fertilizer on the nutrient status and stand growth of Scots pine on drained mires. *Baltic Forestry* 10, 2–10.
- Moilanen, M., Silfverberg, K., Hokkanen, T., 2002. Effects of wood-ash on the tree growth, vegetation and substrate quality of a drained mire: a case study. *For. Ecol. Manage.* 171, 321–338.
- Myhre, G., Shindell, D., Bréon, F.-M., Collins, W., Fuglestedt, J., Huang, J., Koch, D., Lamarque, J.-F., Lee, D., Mendoza, B., Nakajima, T., Robock, A., Stephens, G., Takemura, T., Zhang, H., 2013. Anthropogenic and natural radiative forcing. In: Stocker, T.F., Qin, D., Plattner, G.-K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, P.M. (Eds.), *Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.
- Neubauer, S., Megonigal, P., 2015. Moving beyond global warming potentials to quantify the climatic role of ecosystems. *Ecosystems* 18, 1000–1013. <https://doi.org/10.1007/s10021-015-9879-4>.
- Nieminen, M., 2000. Phosphorus Fertilizer Leaching from Drained Ombrotrophic Peatlands Forests: Empirical Studies and Modelling. Finnish Forest Research Institute, pp. 50 (Research Papers 756).
- Nieminen, M., Penttilä, T., 2004. Inorganic and organic phosphorus fractions in peat from drained mires in northern Finland. *Silva Fennica* 38 (3), 243–251.
- Ojanen, P., Lehtonen, A., Heikkinen, J., Penttilä, T., Minkkinen, K., 2014. Soil CO₂ balance and its uncertainty in forestry-drained peatlands in Finland. *For. Ecol. Manage.* 325, 60–73. <https://doi.org/10.1016/j.foreco.2014.03.049>.
- Ojanen, P., Minkkinen, K., Alm, J., Penttilä, T., 2010. Soil-atmosphere CO₂, CH₄ and N₂O fluxes in boreal forestry-drained peatlands. *For. Ecol. Manage.* 260, 411–421. <https://doi.org/10.1016/j.foreco.2010.04.036>.
- Ojanen, P., Minkkinen, K., Alm, J., Penttilä, T., 2018. Corrigendum to “Soil-atmosphere CO₂, CH₄ and N₂O fluxes in boreal forestry-drained peatlands” [*Forest Ecology and Management* 260: 411–421]. *For. Ecol. Manage.* 412, 95–96. <https://doi.org/10.1016/j.foreco.2018.01.020>.
- Ojanen, P., Minkkinen, K., Lohila, A., Badorek, T., Penttilä, T., 2012. Chamber measured soil respiration: a useful tool for estimating the carbon balance of peatland forest soils? *For. Ecol. Manage.* 277, 132–140. <https://doi.org/10.1016/j.foreco.2012.04.027>.
- Ojanen, P., Minkkinen, K., Penttilä, T., 2013. The current greenhouse gas impact of forestry-drained boreal peatlands. *For. Ecol. Manage.* 289, 201–208. <https://doi.org/10.1016/j.foreco.2012.10.008>.
- Paavilainen, E., 1979. *Metsänlannoitusopas*. Yhteiskirjapaino Oy, Helsinki. pp. 112. ISBN 951-26-1764-1.
- Paavilainen, E., Päivänen, J., 1995. *Peatland Forestry. Ecology and Principles*. In: *Ecol. Stud.* Springer, pp. 248.
- Päivänen, J., Hännel, B., 2012. *Peatland Ecology and Forestry – a Sound Approach*. University of Helsinki, Department of Forest Sciences Publications, pp. 267.
- Pearson, M., Saarinen, M., Minkkinen, K., Silvan, N., Laine, J., 2012. Short-term impacts of soil preparation on greenhouse gas fluxes: a case study in nutrient-poor, clearcut peatland forest. *For. Ecol. Manage.* 283, 10–26. <https://doi.org/10.1016/j.foreco.2012.07.011>.
- Piirainen, S., Domisch, T., Moilanen, M., Nieminen, M., 2013. Long-term effects of ash fertilization on runoff water quality from drained peatland forests. *For. Ecol. Manage.* 287, 53–66. <https://doi.org/10.1016/j.foreco.2012.09.014>.
- Pumpanen, J., Pumpanen, J., Kolari, P., Ilvesniemi, H., Minkkinen, K., Vesala, T., Niinistö, S., Lohila, A., Larmola, T., Morero, M., Pihlatie, M., Janssens, I., Yuste, J.C., Grünzweig, J.M., Reth, S., Subke, J.A., Savage, K., Kutsch, W., Østregren, G., Ziegler, W., Anthoni, P., Lindroth, A., Hari, P., 2004. Comparison of different chamber techniques for measuring soil CO₂ efflux. *Agric. For. Meteorol.* 123, 159–176.
- Reinikainen, A., Veijalainen, H., 1983. Diagnostic use of needle analysis in growth disturbed Scots pine stands. *Comm. Inst. For. Fenn.* 116, 44–48.
- Repola, J., 2008. Biomass equations for birch in Finland. *Silva Fennica* 42 (4), 605–624.
- Repola, J., 2009. Biomass equations for Scots pine and Norway spruce in Finland. *Silva Fennica* 43 (3), 625–647.
- Rütting, T., Björk, R., Meyer, A., Klemetsson, L., Sikström, U., 2014. Reduced global warming potential after wood ash application in drained Northern peatland forests. *For. Ecol. Manage.* 328, 159–166. <https://doi.org/10.1016/j.foreco.2014.05.033>.
- Ruuhijärvi, R., 1983. The Finnish mire types and their distribution. In: Gore, A.J.P. (Ed.), *Ecosystems of the world. 4 B. Mires: swamp, bog, fen and moor. Regional studies*. Elsevier, Amsterdam, pp. 47–67.
- Saarsalmi, A., Mälkönen, E., 2001. Forest fertilization research in Finland: a literature review. *Scand. J. For. Res.* 16, 514–535.
- Saarsalmi, A., Smolander, A., Moilanen, M., Kukkola, M., 2014. Wood ash in boreal, low-productive pine stands on upland and peatland sites: long-term effects on stand growth and soil properties. *For. Ecol. Manage.* 327, 86–95. <https://doi.org/10.1016/j.foreco.2014.04.031>.
- Sarkkola, S., Hökkä, H., Koivusalo, H., Nieminen, M., Ahti, E., Päivänen, J., Laine, J., 2010. Role of tree stand evapotranspiration in maintaining satisfactory drainage conditions in drained peatlands. *Can. J. For. Res.* 40, 1485–1496. <https://doi.org/10.1139/X10-084>.
- Silvola, J., Alm, J., Alholm, U., Nykänen, H., Martikainen, P.J., 1996. CO₂ fluxes from peat in boreal mires under varying temperature and moisture conditions. *J. Ecol.* 84, 219–228.
- Silfverberg, K., Hotanen, J.-P., 1989. Puuntuhan pitkäaikaisvaikutukset ojitetulla mesotrofisella kalvakkanevalla Pohjois-Pohjanmaalla (Summary: Long-term effects of wood-ash on a drained mesotrophic Sphagnum papillosum fen in Oulu district, Finland). *Folia Forestalia* 742, 1–23.
- Silfverberg, K., Issakainen, J., Moilanen, M., 2011. Growth and nutrition of Scots pine on drained and fertilized purple moor grass fens in central Finland. *Baltic Forestry* 17 (1), 91–101.
- Simola, H., Pitkänen, A., Turunen, J., 2012. Carbon loss in drained forestry peatlands in Finland, estimated by re-sampling peatlands surveyed in the 1980s. *Eur. J. Soil Sci.* 63, 798–807. <https://doi.org/10.1111/j.1365-2389.2012.01499.x>.
- Uri, V., Kukumägi, M., Aosaar, J., Varik, M., Becker, H., Morozov, G., Karoles, K., 2017. Ecosystems carbon budgets of differently aged downy birch stands growing on well-drained peatlands. *For. Ecol. Manage.* 399, 82–93. <https://doi.org/10.1016/j.foreco.2017.05.023>.
- Vasander, H., 1982. Plant biomass and production in virgin, drained and fertilized sites in a raised bog in southern Finland. *Ann. Bot. Fenn.* 19, 103–125.
- Veijalainen, H., 1983. Geographical distribution of growth disturbances in Finland. *Comm. Inst. For. Fenn.* 116, 13–16.
- Wilson, D., Blain, D., Couwenberg, J., Evans, C., Murdiyarso, D., Page, S., Renou-Wilson, F., Rieley, J., Sirin, A., Strack, M., Tuittila, E.-S., 2016. Greenhouse gas emission factors associated with rewetting of organic soils. *Mire Peat* 17, 4. <https://doi.org/10.19189/Map.2016.OMB.222>.