Biogeosciences Discuss., 10, 5107–5148, 2013
www.biogeosciences-discuss.net/10/5107/2013/
doi:10.5194/bgd-10-5107-2013

Biogeosciences
Discussions
Open Access

A fertile peatland forest does not constitute a major greenhouse gas sink

A. Meyer[1], L. Tarvainen[1], A. Nousratpour[1], R. G. Björk[2], M. Ernfors[3], Å. Kasimir Klemedtsson[2], A. Lindroth[4], M. Räntfors[1], T. Rütting[2], G. Wallin[1], P. Weslien[2], and L. Klemedtsson[2]

[1]Department of Biological and Environmental Sciences, University of Gothenburg, Sweden
[2]Department of Earth Sciences, University of Gothenburg, Sweden
[3]Department of Agrosystems, Swedish University of Agricultural Sciences, Alnarp, Sweden
[4]Department of Physical Geography and Ecosystems Science, Lund University, Sweden

Received: 6 February 2013 – Accepted: 3 March 2013 – Published: 13 March 2013

Correspondence to: A. Meyer (astrid.meyer@yahoo.de)

Published by Copernicus Publications on behalf of the European Geosciences Union.

Abstract

Afforestation has been proposed as a strategy to mitigate the often high greenhouse gas (GHG) emissions from agricultural soils with a high organic matter content. However, the carbon dioxide (CO_2) and nitrous oxide (N_2O) fluxes after afforestation can be considerable, depending predominantly on site drainage and nutrient availability. Studies on the full GHG budget of afforested organic soils are scarce and hampered by the uncertainties associated with methodology. In this study we determined the GHG budget of a spruce-dominated forest on a drained organic soil with an agricultural history. Two different approaches for determining the net ecosystem CO_2 exchange (NEE) were applied: for the year 2008, direct (eddy covariance) and an indirect (analyzing the different components of the GHG budget), so that uncertainties in each method could be evaluated. The annual tree production in 2008 was 8.2 (± 1.7) $\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$ due to the high levels of soil nutrients, the favorable climatic conditions and the fact that the forest was in its optimum growth phase. N_2O fluxes were determined by the closed chamber technique and amounted to 3.3 (± 2.4) $\mathrm{t\,CO_{2eq}\,ha^{-1}\,yr^{-1}}$. According to the direct measurements from the eddy covariance technique, the site acts as a minor GHG sink of −4.1 (± 2.6) $\mathrm{t\,CO_{2eq}\,ha^{-1}\,yr^{-1}}$. This contrasts with the NEE estimate derived from the indirect approach which suggests that the site is a net GHG emitter of 3.3 (± 10.1) $\mathrm{t\,CO_{2eq}\,ha^{-1}\,yr^{-1}}$. Irrespective of the approach applied, the soil CO_2 effluxes counter large amounts of the C sequestration by trees. Due to major uncertainties involved in the indirect approach, the direct approach is considered the more reliable tool. As the site was in its optimum growth stage, i.e. the rate of C sequestration was at its maximum and will decrease with forest age, it will probably become a GHG source once again as the trees mature. Since forests in their younger stages are usually GHG sources or have no effect on GHGs, the overall sink potential of this afforested nutrient-rich organic soil is probably limited to only a short period.

1 Introduction

After drainage, organic soils continuously lose CO_2 to the atmosphere, irrespective of their use as agricultural or forest land. Hence, such soils contribute to global warming (Smith and Conen, 2004; Davidson and Janssens, 2006; Couwenberg et al., 2011). The soil CO_2 emissions from drained organic forest soils are reported within the Land Use and Land Use Change and Forestry (LULUCF) sector for UNFCCC and Kyoto submissions (http://unfccc.int/national_reports/annex_i_ghg_inventories/items/2715.php). Within the LULUCF sector, emissions from drained organic soils are also reported, but these emissions are not made visible because they constitute only a value to be subtracted from the uptake of CO_2 by growing forests. However, the CO_2 emissions from drained organic soils are comparable in size to the Swedish industrial sector, including mining and steel manufacturing (UNFCCC, National Inventory Reports (NIR): http://unfccc.int/national_reports/annex_i_ghg_inventories/national_inventories_submissions/items/6598.php). Moreover, N_2O emissions are not reported at all for drained organic forest soils even though they can be considerable (Alm et al., 2007; Ernfors et al., 2008).

It has been estimated that organic soils used for agriculture release an amount comparable to 6–10 % of the total reported Swedish GHG emissions (Kasimir Klemedtsson et al., 1997; Berglund and Berglund, 2010), despite the fact that only 5–10 % of agricultural land is on such soils. Such emissions are primarily a result of drainage, which increases soil aeration and consequently organic matter mineralization and results in the release of carbon (C) and nitrogen (N) as atmospheric trace gases (e.g. Kasimir Klemedtsson et al., 1997; Maljanen et al., 2010). The main GHG is CO_2, with reported emissions of 3.2–8.3 $\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$ (Nykänen et al., 1998; Maljanen et al., 2004, 2007), but N_2O emissions can also be high (2–37 $\mathrm{kg\,N\,ha^{-1}\,yr^{-1}}$: Höper et al., 2002; Kasimir Klemedtsson et al., 2009). Methane (CH_4) emissions from drained organic soils are usually of limited importance (Huttunen et al., 2003; Maljanen et al., 2003a; von Arnold et al., 2005; Alm et al., 2007; Kasimir Klemedtsson et al., 2009). Afforestation has been discussed as a mitigation strategy to reduce GHG emissions from cropland soils because it is assumed that the reduction in fertilizer applications and management activities, and especially the increase in plant C accumulation will convert the ecosystem into an overall GHG sink (Watson et al., 2000; Hargreaves et al., 2003). In Sweden, the area of drained agricultural organic soils (cropland and pasture) decreased from 0.7 Mha in 1947 to 0.3 Mha in 2003 (Berglund and Berglund, 2010), and most of this area has probably been converted to forestland. Drained organic soils used for forestry currently cover about 1.1 Mha (NIR Sweden; von Arnold, 2004).

Forests on drained organic soils have long been assumed to constitute an overall GHG sink, even when the soil acts as a C source; this is because the C accumulation by the trees is assumed to compensate for soil losses (Minkkinen et al., 1999, 2002; Hargreaves et al., 2003). The Swedish NIR (2012) submitted to the UNFCCC uses emission factors (EFs) for CO_2 of 3.0 and 1.9 $\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$ for well drained and poorly drained forest soils, respectively, based on the work of von Arnold (2004). With respect to N_2O, the IPCC Good Practice Guidelines (IPCC GPG 2003: http://www.ipcc-nggip.iges.or.jp/public/gpglulucf/gpglulucf_contents.html) only consider it necessary to report emissions from wet, usually poorly-drained or undrained, forest soils (GPG 2003 Appendix 3a.2), so well-drained organic soils are ignored. Previous studies, however, indicate that drained organic soils can be responsible for considerable CO_2 and N_2O emissions for several years after afforestation (CO_2: up to 5.2 $\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$, Alm et al., 2007) because the soil fertility, the abundant availability of C substrates, and the low water table favor microbial decomposition (Maljanen et al., 2003a; von Arnold et al., 2005); this means that high soil C and N losses outweigh C accumulation by the trees (Alm et al., 2007). As a consequence, the calculation of GHG budgets for drained organic forest soils based on IPCC GPG (2003) is associated with great uncertainty. In addition, existing studies have rarely taken emissions of all GHGs at a site into account, so that there is much uncertainty with regard to the estimation of GHG budgets based on empirical data (Maljanen et al., 2010).

To our knowledge only Lohila et al. (2011) have investigated all major GHGs (CO_2, N_2O, CH_4) from a forest on a drained organic soil for a full year, although short-term studies with specific focus on frost events and clear-cutting also exist (Mäkiranta et al., 2010; Pihlatie et al., 2010). Using micrometeorological measurements, Lohila et al. (2011) found that a pine forest on a drained organic soil was an intermediate C sink, in which N_2O and CH_4 only had a minor impact on the total GHG budget. However, this site had not been in agricultural use previously and had a high soil C : N ratio (34–41), so low N_2O emissions could be expected (Klemedtsson et al., 2005).

The soil N content, which is often high in former agricultural soils, seems to be a key indicator of GHG emissions (Minkkinen et al., 1999). Klemedtsson et al. (2005) found a clear correlation between soil C : N and N_2O emissions (Klemedtsson et al., 2005; Mäkiranta et al., 2007; Ernfors et al., 2008), and this was subsequently used by Ernfors et al. (2008) to extrapolate N_2O emissions to the country level. They found that afforested organic soils are responsible for 15 % to the total Swedish anthropogenic N_2O emissions, although this figure has not yet been used in the UNFCCC reports. However, the release of N due to mineralization is primarily governed by the soil water level, so that the drainage status of the soil is the determining factor (Martikainen et al., 1993). Therefore, drained nutrient-rich afforested organic soils are potentially large sources of GHGs.

The GHG budgets of forests on drained organic soils are uncertain due to both the scarcity of studies and the uncertainties associated with existing methods for determining the GHG budget. The most frequently used indirect approach is to quantify the soil GHG fluxes using the chamber method and to relate these to the biomass production of the trees as determined by biomass functions (e.g. Marklund et al., 1989; Minkkinen et al., 2001; Jalkanen et al., 2005). The chamber technique, however, has been shown to give different results depending on the equipment and chamber design used (Pumpanen et al., 2004; Christansen et al., 2011). Despite this, a recent study by Ojanen et al. (2012) on the C balance of peatlands identifies the chamber technique as a useful tool for GHG inventories, but only when the associated uncertainties are tolerable. The NEE can only be determined by the use of micrometeorological methods (Mäkiranta et al., 2007; Lohila et al., 2007, 2011). A draw-back of this method is that measurements are laborious to set-up and cost-intensive. Furthermore, the technique does not work well under highly stable atmospheric conditions (~~Moffat et al., 2007;~~ Vickers et al., 2011).

In view of the need to quantify further the GHG budget of forests on nutrient-rich drained organic soils, we investigated the major GHG fluxes of a drained organic soil in southwestern Sweden. The site has previously been used for agriculture but has been afforested with spruce, pine and birch for about 60 yr. The GHG budget was determined by estimating the ecosystem fluxes of N_2O, and CH_4 and the NEE. To quantify the NEE, we took two different approaches: direct measurement using the eddy covariance technique (EC) and an indirect approach using the chamber technique and flux balance calculations. This allowed us to compare the two approaches with regard to their limitations and uncertainties and to analyze the effect on the overall budget.

2 Material and methods

2.1 Site description

The study was conducted in a forest stand on a drained organic soil in the Skogaryd research catchment (58°23′ N, 12°09′ E) in southwest Sweden (Klemedtsson et al., 2010). The former mesotrophic fen was drained in the 1870s and was used for agriculture until 1951 when it was afforested. During the study period, the stand density was approximately 1000 trees ha^{-1}. Norway spruce, *Picea abies* (L.) Karst., dominated the stand (82 % of BA), but Scots pine, *Pinus sylvestris* L. (13 % of BA) and Silver birch, *Betula pendula* Roth (5 % of BA), were also present. The field vegetation was dominated by bryophytes with sparse vascular plants, including species such as *Luzula pilosa* and *Oxalis acetosella*. The soil was classified as a Histosol (IUSS 2006) with the organic layer more than 1 m deep and a C content of 116 ± 16 t C ha^{-1} in the upper 50 cm. The average soil pH in the upper 20 cm was between 4.4 and 4.6 (Ernfors, 2009). Soil properties were determined in October 2007, the soil organic matter content was 85 % in the upper 15 cm and 87 % at a depth of 15–40 cm. The site is well drained so that the average water table in 2008 was about 38–41 cm in winter and 63.3 cm in summer (May to September). The soil C : N ratio ranged between 22.7 and 26.8 in the upper 50 cm.

The 30-yr (1961–1990) annual mean temperature at the Vänersborg weather station, situated approximately 10 km east of the site, was 6.2 °C. The 30-yr annual mean precipitation at the closest weather stations at Vänersborg and Uddevalla (12 km west of the site) was 709 mm and 870 mm, respectively (http://www.smhi.se). The precipitation at the Skogaryd site in 2008 was found to correlate strongly ($r^2 = 0.98$) with the mean of the two weather stations and we therefore assume that the long-term annual mean precipitation at the site is approximately 785 mm.

The position of the eddy covariance tower as well as the soil chambers is indicated in Fig. 1a. The forest stand surrounding the flux tower is relatively homogeneous and extends about 200 m in all directions, except towards the E and SE. The prevailing wind is from the SW (Fig. 1b) where the drained afforested organic site extends for more than 300 m.

2.2 Measurement of abiotic variables

To support the measurements of GHG fluxes, data on environmental parameters were collected. Air temperature was measured at the EC tower 1.5 m above the ground. Soil temperature and soil moisture were recorded every 30 min and measured by means of soil sensors, buried in six plots within the study area installed at soil depths of 0.05, 0.15, 0.30 and 0.60 m. Soil moisture was measured with TDR sensors (see Table A1) and a site-specific calibration was applied since the standard calibration is not applicable to organic soils. Details of the instruments used are given in Table A1.

Title Page
Abstract Introduction
Conclusions References
Tables Figures
◀I ▶I
◀ ▶
Back Close
Full Screen / Esc
Printer-friendly Version
Interactive Discussion

2.3 Determination of GHG budgets

To determine the ecosystem budget of major GHGs for the year 2008, the net flux of CO_2, N_2O and CH_4 was determined and transformed into CO_2 equivalents based on the 100 yr global warming potential (GWP) of each gas. The net flux of N_2O and CH_4 was determined from soil efflux measurements, whereas the NEE was determined using two approaches. In the direct approach (1), NEE was determined by continuous measurement of the CO_2 flux over the forest canopy using the eddy covariance (EC) technique. In the indirect approach (2), the NEE was quantified by measuring compartmental fluxes and calculating the NEE using the following equations:

$$\mathrm{NEE} = \mathrm{GPP} - R = \mathrm{GPP} - R_\mathrm{a} - R_\mathrm{h} \tag{1}$$

$$\mathrm{NPP} = \mathrm{GPP} - R_\mathrm{a} \tag{2}$$

where GPP is gross primary production, NPP is net primary production, R is total respiration, R_a is autotrophic respiration and R_h is heterotrophic respiration. Combining Eqs. (1) and (2) gives:

$$\mathrm{NEE} = \mathrm{NPP} - R_\mathrm{h} \tag{3}$$

Biomass production during one year (ΔB) can be derived from:

$$B_1 = B_0 + \mathrm{NPP} - L \tag{4}$$

$$\Delta B = B_1 - B_0 = \mathrm{NPP} - L \tag{4a}$$

where B_0 and B_1 are total biomass at the beginning and end of the year, respectively, and L is the total litter production. Equation (4a) can be rearranged as follows:

$$\mathrm{NPP} = \Delta B + L = \mathrm{AGB_{inc}} + \mathrm{BGB_{inc}} + L_\mathrm{L} + L_\mathrm{R} \tag{4b}$$

where $\mathrm{AGB_{inc}}$ and $\mathrm{BGB_{inc}}$ are the net above- and belowground biomass increments, respectively, and L_L and L_R are the litter production from leaves (plus branches etc.)

and roots, respectively. Furthermore, heterotrophic respiration (R_h) can be partitioned into respiration from decomposition of soil organic matter (R_{SOM}) and decomposition of leaf (plus branches etc.) and root litter (R_{LL}, R_{LR}):

$$R_h = R_{SOM} + R_{LL} + R_{LR} \tag{5}$$

Combining Eqs. (3), (4b) and (5) gives:

$$\mathrm{NEE} = \mathrm{AGB_{inc}} + \mathrm{BGB_{inc}} + L_L + L_R - (R_{SOM} + R_{LL} + R_{LR}) \tag{6}$$

In a closed canopy forest we can assume that the annual decomposition of litter from both leaves and roots equals the annual litter production by leaves and roots. For such a forest in a quasi steady state, Eq. (6) can be simplified to:

$$\mathrm{NEE} = \mathrm{AGB_{inc}} + \mathrm{BGB_{inc}} - R_{SOM} \tag{6a}$$

So that

$$R_{SOM} = \mathrm{AGB_{inc}} + \mathrm{BGB_{inc}} - \mathrm{NEE} \tag{6b}$$

Equation (6a) is used in approach 2 to calculate the NEE, based on determination of $\mathrm{AGB_{inc}}$, $\mathrm{BGB_{inc}}$ and R_{SOM}, while in approach 1, based on eddy flux measurements of the NEE, R_{SOM} was calculated according to Eq. (6b). Details of the methods associated with each approach are given below.

2.4 Micrometeorological measurements of NEE (approach 1)

During 2008 continuous measurements of the CO_2 flux above the canopy were carried out using the EC technique (Grelle and Lindroth, 1996). The instrumentation used was broadly the same as the system described in Lindroth et al. (2008), and only the differences are noted here. Instrumentation was mounted on a scaffold tower. Air temperature and air humidity were measured at a height of 22 m, while global radiation, net radiation and photosynthetically active radiation (PAR) were measured at the top of the scaffold tower at 27 m, where the EC system was located (see also Table A1). The air intake for the gas analyzer was mounted on a vertical pole which extended about 3 m above the top of the scaffold. The air for concentration analysis was drawn into the gas analyzer by means of a membrane pump at a rate of 10 L min^{-1} through a 5 m long 4 mm inner diameter polyethylene tube. For calibration of the CO_2 analyzer, an automatic 2-point system was used fortnightly, but was replaced by manual calibration later in the year. The energy balance closure was estimated by the annual flux sums and was about 90 %.

Data collection and analysis were conducted in real time by the Ecoflux software (InSituFlux AB, Ockelbo, Sweden). Post-collection data analysis and gap filling followed the method described by Lindroth et al. (2008). For this, the vertical CO_2 gradient was measured at seven different heights (0.05, 1.5, 10, 15, 18, 22 and 25 m above the ground) using a custom made gas exchange system (see Wallin et al., 2001 for details). In the first step, data were selected for periods when the flux measurement systems worked properly and spikes were removed which could not be attributed to morning flushes. The CO_2 storage flux was calculated using data from the measurements of the vertical CO_2 gradient added to all remaining flux data to obtain the biotic flux.

The gap filling was conducted as follows: Daytime and nighttime data were selected according to the $u^* >$ threshold, i.e. the friction velocity at which the eddy covariance criteria are fulfilled. The nighttime flux, including storage, was plotted against u^* and the u^* threshold value determined visually as the point where the normalized flux started to decrease; this had a value of 0.25 m^{-2} s^{-1}.

Half-hourly data were grouped into 14 day periods and the biotic flux and PAR were fitted to the function:

$$F_c = -(F_{csat} + R_d) \cdot \left[1 - \exp\left(\frac{-\alpha \mathrm{PAR}}{F_{csat} + R_d}\right)\right] + R_d \tag{7}$$

where F_c is the 30-min daytime biotic flux and PAR is the photosynthetically active radiation (µmol m^{-2} s^{-1}). F_{csat}, R_d and α are fitting parameters.

Nighttime data from each period were fitted to the function:

$$F_{\mathrm{cn}} = R_0 \cdot \exp(k \cdot T) \tag{8}$$

where F_{cn} is the nighttime biotic flux ($\mu\mathrm{mol\,m^{-2}\,s^{-1}}$), R_0 and k are fitting parameters and T is the air temperature (K).

If no relationship with temperature could be obtained for nighttime data, the mean value during the night for the whole period was used. For longer periods, mean diurnal half hourly look-up tables were created using existing data based on at least 10 values per half hour (Lindroth et al., 2008). In total, 12 % of the annual flux had to be gap-filled due to loss of data, a further 30 % had to be gap-filled because they did not meet the u^* criterion. The flux footprints were estimated by the Eddy-Pro software (Li-Cor Inc., Lincoln, USA). The primary choice was the footprint model of Kljun et al. (2004), but during stable conditions when this model was not applicable, the Kormann and Meixner (2001) model was used.

2.5 Quantification of above and belowground tree growth

The annual production of aboveground and belowground biomass (including roots ≥ 1 cm) (Eq. 6a, b: $\mathrm{AGB_{inc}}$, $\mathrm{BGB_{inc}}$) was determined using biomass functions developed by Marklund (1988) and modified by Minkkinen et al. (2001) for organic soils; these require diameter at breast height (dbh) and tree height as input parameters. These parameters were measured for 170 trees from 16 sub plots of 0.01 ha distributed in a circle around the EC tower.

Diameter at breast height was measured at the end of September/beginning of October in 2007 and 2008. The height of these trees was estimated in 2007 using Korf's function which assumes an exponential relationship between dbh and tree height (Mehtätalo, 2004). Korf's function was parameterized from measurements of dbh and tree height in November 2007 on approximately 335 trees in a 90 × 90 m plot surrounding the EC tower. The tree height increment in the 16 sub-plots was calculated from the mean annual increment in height, calculated as:

$$h_{\text{inc}} = \frac{h_{\text{tot}} - 1.3}{a_{\text{tot}} - a_{\text{dbh}}} \tag{9}$$

where h_{inc} = mean annual height increment (m), h_{tot} = total tree height (m), a_{tot} = total age (yr), and a_{dbh} = age at breast height (yr).

The age at breast height was estimated from wood cores. The method was shown to be valid for the height increment of the trees near the tower for the last 10 yr. Based on wood analysis, 49 % was used as the conversion factor for C content in dry biomass.

In 2010 additional measurements of dbh and basal area (BA) were taken from trees within an area of 0.81 ha across the study site. In order to compare the site to the small plots of 0.01 ha, BA was re-calculated for 2008 (assuming an annual BA increase of 1.1 $\text{m}^2\,\text{ha}^{-1}$, Klemedtsson et al., 2010). These data showed that the average BA of the small plots was 23 % lower than the BA of the large plot. Applying a regression analysis, the variation in BA could explain about 50 % of the variation in tree growth, so that the relationship between annual biomass production and BA was used to correct for the, on average, 23 % smaller BA in the 16 subplots compared to the larger area around the EC-tower.

Since fine roots (< 1 cm) were not considered in the biomass functions by Minkkinen et al., (2001), we assumed that annual fine root production corresponded to the increment in needle and twig biomass, as determined by the Minkkinen et al. (2001) equations; this amounted to 3.6 % of the total biomass increment.

2.6 Soil respiration (approach 2)

The quantification of the respiration associated with the soil organic matter (SOM) (R_{SOM}, Eq. 6b) was based on measurements of the soil CO_2 efflux (E_{CO_2}) which were determined by the chamber method and a trenching experiment. Details of the trenching experiment are described by Ernfors et al., (2011). Decaying roots and mycelia, cut by the trenching, added to the measured CO_2 flux from SOM so that the measured

E_{CO_2} originates from three different sources:

$$E_{CO_2} = R_{SOM} + R_{decay} + R_{LL} \tag{10}$$

So that

$$R_{SOM} = E_{CO_2} - R_{decay} - R_{LL} \tag{10a}$$

where R_{SOM} and R_{LL} are the respiration from the decomposition of SOM and litter ($t\,C\,ha^{-1}\,yr^{-1}$) (see Eq. 5) and R_{decay} ($t\,C\,ha^{-1}\,yr^{-1}$) is the respiration from decaying roots and mycelia which were cut by trenching.

E_{CO_2} was measured using automatic transparent closed chambers at three stations within the footprint area of the EC tower (Fig. 1a). Chambers were placed on permanently installed collars that covered the trenching frames. All vegetation was removed to eliminate CO_2 uptake by photosynthesis. During measurement, the lid was closed and air circulated between an infrared gas analyzer and the chamber. To ensure homogenous mixing of the air, chambers were equipped with a fan. A logger system connected to the chambers prompted a measurement cycle every 20 min, during which the CO_2 concentration was measured every 30 s for 15 min. Chambers were rotated every 2–3 weeks between stations.

The additional flux from decomposing trenched roots and mycelia, R_{decay}, was determined by a modified version of the method of Ngao et al. (2007) and approximated biomass decay over time (see e.g. Berg and McClaugherty, 2008). It describes an exponential decay of root biomass after trenching within the first year. Originally developed for fine roots, we extended Ngao's functions to account for coarse roots and fungal mycelia by adapting the function parameters:

$$R_{decay} = (1-\alpha)\cdot m\cdot(1-\exp(-0.997\cdot k_r)) \tag{11}$$

where α = microbial efficiency of decomposition, m = biomass of coarse roots, fine roots and mycelia ($t\,C\,ha^{-1}$), and k_r = decomposition constant.

The root biomass was determined by soil coring on 5 December 2007 using a peat sampler (with a rectangular cross section measuring 8.2 × 8.3 cm). Two soil cores per plot ($n = 6$) were taken to a depth of 40 cm. The amount of extramatrical mycorrhizal mycelia was estimated using the mesh bag technique (see Wallander et al., 2012 for a detailed description).

The CO_2 flux from the litter layer, R_{LL} that contributes to the measured soil efflux E_{CO_2} (Eq. 10) was determined as the difference between the annual aboveground litter production from trees (total input to litter layer) and the average annual accumulation of C in the litter layer, assuming long-term equilibrium between inputs and outputs:

$$R_{LL} = L_L - \Delta L \tag{12}$$

where L_L = average annual aboveground litter production ($\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$), and ΔL = average annual accumulation of C in the litter layer ($\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$).

In order to estimate ΔL, a total of 30 litter layer samples were taken across the site using a metal auger. After drying for 24 h at 70 °C, the annual accumulation was determined as the total mass of C divided by the number of years since the start of litter layer formation, i.e. since afforestation.

Aboveground plant litter production, L_L, was quantified by traps made of nylon mesh ($0.25\,\mathrm{m^2}$; $n = 10$) that were placed evenly across the study area. Traps were emptied every sixth week and all litter fractions (needles, twigs, cones, branches) were considered and the C content determined from sub-samples.

2.7 Soil N_2O and CH_4 fluxes

Besides the CO_2 fluxes described above, N_2O and CH_4-fluxes were quantified to determine the total GHG budget of the study site. Measurements of the N_2O and CH_4 fluxes were carried out once every two weeks using the closed chamber method and are described in detail by Ernfors et al. (2011). Gas samples from the chambers were taken 4, 8, 16 and 32 min. after lid closure. Gas concentration was analyzed by gas chromatography and the gas flux was determined from the slope of the regression line of the gas concentration plotted against time. Only N_2O and CH_4 fluxes from control plots (non-trenched plots) were used in the subsequent calculations.

2.8 Handling of the flux data and calculation of annual flux rates

Following the standard convention, we use a negative sign to indicate uptake and a positive sign to indicate gas release from the ecosystem. CH_4 and N_2O fluxes were averaged for each station for the fortnightly measurements and then interpolated between measurements and summed for the whole year. The same procedure was applied to the 2- to 3- week periods when respiration was measured. Standard deviation is given as the variation between stations 1, 2 and 3. Annual fluxes of N_2O and CH_4 were converted to CO_2-equivalents using the GWP for a 100 yr time frame of 298 (N_2O) and 25 (CH_4) (IPCC: Forster et al., 2007).

The propagated random error of $\mathrm{NEE}_{\mathrm{calc}}$ and R_{SOM} was calculated according to the Gauss error propagation scheme, assuming that the respective components were independent:

$$\mathrm{SE}_{\mathrm{tot}} = \sqrt{\mathrm{SE}_{c1}^2 + \mathrm{SE}_{c2}^2 + \mathrm{SE}_{cx}^2} \tag{13}$$

where $\mathrm{SE}_{\mathrm{tot}}$ = propagated error, and SE_{cx} = error of component $1\ldots x$.

3 Results

3.1 Abiotic data

The year 2008 was considerably warmer (8.7 °C) than the 30-yr average for Vänersborg (6.4 °C). Records from the EC tower in Skogaryd showed that early winter (January/February) air temperature rarely fell below 0 °C and just once in March it reached −7 °C (Fig. 2a). From the beginning of May, the daily mean air temperature was always well above 10 °C apart from a short period in late May. The highest daily mean temperatures were recorded in late July (23.9 °C). Temperature dynamics of the second winter period 2008 (November/December) were similar to the first period (January/February) and frost only occurred occasionally. The precipitation during 2008 at Vänersborg was above average (10-yr average: 709 mm, 2008: 1006 mm) and relatively evenly distributed over the year, except during spring (April to June) when rainfall was 30 % less (127 mm) than the 10-yr average for that period (177 mm). A persistent snow layer of up to 35 cm was observed for only two weeks in March (http://www.smhi.se).

3.2 NEE as determined by micrometeorological methods

During daytime, the source distance distribution around the tower peaked, in general, at 100–200 m and the 90 % limit was at 300–350 m. During nighttime, the source area occasionally moved further away. This, in relation to homogeneous fetch conditions, means that the measured fluxes were certainly dominated by the surrounding forest stand as the prevailing wind direction was S or SW (Fig. 1b).

During the first winter period NEE ranged between 0 and 1.5 $\mathrm{g\,C\,m^{-2}\,d^{-1}}$ with a maximum release of 2.3 $\mathrm{g\,C\,m^{-2}\,d^{-1}}$. The NEE was positive from 1 January until 29 March when the ecosystem switched to being a C sink (Fig. 2b). NEE decreased rapidly until mid-May but then remained constant until the end of June. The highest net CO_2 uptake was reported in August (–5.3 $\mathrm{g\,C\,m^{-2}\,d^{-1}}$). In late September, the site returned to acting as a C source, with emissions ranging between 0 and 1.5 $\mathrm{g\,C\,m^{-2}\,d^{-1}}$. In November and December the NEE showed a strong correlation to air temperature ($r^2 = -0.78$). The accumulated NEE in 2008 was –2.0 $\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$.

3.3 Biomass estimations

Standing biomass (including roots ≥ 1 cm) calculated by the Minkkinen et al. (2001) approach amounted to 258 and 271 $\mathrm{t\,dw\,ha^{-1}}$ in 2007 and 2008, respectively. The coarse root biomass (> 1 cm) was estimated to be 45.7 $\mathrm{t\,dw\,ha^{-1}}$ (2007) and 47.9 $\mathrm{t\,dw\,ha^{-1}}$

(2008). The resulting tree growth in 2008, corrected according to the BA factor, was 16.2 $\mathrm{t\,dw\,ha^{-1}\,yr^{-1}}$, equivalent to 7.9 $\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$ excluding fine roots. Including fine roots, the total net tree growth became 8.2 $\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$ for 2008.

3.4 NEE as determined by compartment fluxes

The measured soil CO_2 efflux, E_{CO_2}, in 2008 was 13.1 (± 1.9) $\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$ ~~(Nousratpour, unpublished data)~~. The CO_2 production from decaying roots in the trenched plots was estimated to amount to 1.5 (± 0.3) $\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$ for fine roots and 0.7 (± 0.5) $\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$ for coarse roots, resulting in a total R_{decay} of 2.2 (± 0.4) $\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$ (Table 1). The biomass of mycorrhizal fungal mycelia was negligible, so that the decay from trenched mycelia could be ignored. The average annual litter fall was 2.9 (± 1.1) $\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$ and the annual plant litter accumulation was 0.2 (± 0.1) $\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$, which sums to an annual net CO_2 flux from decomposing litter of 2.7 (± 1.0) $\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$. As a result, the respiration from decomposing SOM, R_{SOM}, was 8.2 (± 2.2) $\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$.

3.5 N_2O and CH_4 fluxes

The measured CH_4 flux was, on average, −4.4 (± 2.2) $\mathrm{kg\,CH_4\,ha^{-1}\,yr^{-1}}$, i.e. CH_4 was taken up by the soil. In contrast, N_2O fluxes constituted a major greenhouse gas source, averaging 11.0 (± 8.0) $\mathrm{kg\,N_2O\,ha^{-1}\,yr^{-1}}$ (see also Ernfors et al., 2011); this corresponds to 3.3 $\mathrm{t\,CO_{2eq}\,ha^{-1}\,yr^{-1}}$ (Table 2).

3.6 Two approaches to determining NEE

The measurements conducted in this study allow a comparison of the two different approaches to quantify the NEE of an afforested organic soil: direct measurement by micrometeorological methods and indirect quantification via soil respiration and C accumulation in tree biomass. The two approaches are illustrated in Fig. 3. The direct approach identified the site as a C sink of −2.0 (± 0.3) t C ha^{-1} yr^{-1} (NEE_{meas}). Since the understorey vegetation is sparse and, hence, exhibits negligible fluxes, the difference between the annual tree growth and NEE_{meas} gives an estimate of the respiration from SOM (R_{res}) of 6.2 t C ha^{-1} yr^{-1}. In the second approach, the NEE is estimated indirectly (NEE_{calc}) based on the difference between annual tree growth and measured heterotrophic soil efflux, corrected for the CO_2 efflux from decaying roots, mycorrhizal fungi and litter. The respiration from SOM (R_{SOM}) was 8.2 t C ha^{-1} yr^{-1} so that the NEE_{calc} of the site was 0.0 (± 2.3) t C ha^{-1} yr^{-1} (Fig. 3).

4 Discussion

4.1 Biomass estimations

Both approaches used in this study to quantify the NEE of a nutrient-rich afforested organic site rely on correctly estimating tree growth on the basis of biomass equations. Uncertainties involved in the application of such equations have been pointed out by Jenkins et al. (2003). In our study, the application of the Minkkinen et al. (2001) equations can be considered as comparatively reliable. First, the equations were specifically developed for Swedish forests, i.e. for the study species, site fertility and site climate. Second, the Minkkinen et al. (2001) equations account for the effect of the soil type (organic) on root growth, which has been shown to be higher in organic than in mineral soils (Laiho and Finer, 1996).

The estimated tree growth of 8.2 t C ha^{-1} yr^{-1} is higher than reported for similar ecosystems in Finland (Table 3). The study site at Skogaryd is very nutrient-rich due to its former use for agricultural crops. This, together with the fact that the site is well drained and located in the hemiboreal zone with a more favorable climate for tree growth, allows for a higher productivity compared to Finnish sites (Lohila et al., 2007, 2011, see Table 3). The lower annual tree growth observed on the nutrient-rich organic soil reported by von Arnold et al. (2005) can probably be explained by tree age. The trees at their site were 90 yr old, and so were already older than the age associated with maximum biomass production (Black and Gallagher, 2010).

4.2 Quantification of the NEE

The two approaches used to quantify the NEE gave different results (approach 1: $-2.0\,(\pm 0.3)\,\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$, approach 2: $0.0\,(\pm 2.3)\,\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$). Due to measurement uncertainties, the measured and estimated NEE are actually not different from each other, even though ~~that~~ the absolute difference is $2\,\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$. The uncertainty associated with $\mathrm{NEE_{calc}}$ is much higher than the uncertainty associated with $\mathrm{NEE_{meas}}$, although it must be remembered that the propagated error of $\mathrm{NEE_{calc}}$ was determined according to the Gauss' error propagation scheme (Eq. 13). Due to the nature of this calculation, a low standard error of the components of the equation will yield a relatively high propagated error estimate.

Comparison of the NEE with that at other sites is hampered because of the scarcity of year-round measurements in the literature. In Finland, Lohila et al. (2011) found a NEE of $-2.4\,\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$ for a pine forest on a drained organic soil (Table 3). Not having an agricultural history, this soil had a much lower soil N content (C : N of 34 to 41) leading to lower C accumulation in trees and, in contrast to the present site, a net accumulation of C in the soil due to limited peat decomposition. A young forest on former agricultural land was found to be a ~~restricted~~ C source of $0.5\,\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$ (Lohila et al., 2007). In the latter study, the organic soil lost large amounts of C that could not be completely compensated for by the C accumulation in the trees. However, the forest was only 30 yr old with a C accumulation of $2.4\,\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$ which did not include fine roots. Furthermore, the C balance at that site could not be quantified completely and Lohila et al. (2007) stated that C accumulation in ground vegetation and belowground litter production could be possible additional C sinks. Thus, the actual C accumulation was probably higher and since C sequestration into trees is likely to increase as long as the canopy is not closed, the site ~~will probably switch to being~~ a C sink in the future (Hargreaves et al., 2003; Black and Gallagher, 2010). Other studies on the NEE of forests on organic soils have been undertaken (e.g. Dunn et al., 2007; Flanagan and Syed, 2011), but comparisons are of limited value because site conditions differ considerably with respect to vegetation, management history and soil properties. Overall, the $\mathrm{NEE_{meas}}$ from our study appears to be reasonable compared to values reported from Finland, whereas the $\mathrm{NEE_{calc}}$ seems comparatively high considering the high level of C accumulation in trees of $8.2\,\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$. The respiration from SOM as determined using the chamber technique (R_{SOM}) was about 30 % higher compared to the residual flux R_{res} for approach 1; this may be because of uncertainties in the respective methods. R_{SOM} is at the upper end of the range of reported values from organic sites across Finland and Sweden (-0.8 to $6.7\,\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$, see Table 3) and even higher than fluxes from nutrient rich agricultural sites (4.5 to $5.5\,\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$, Kasimir Klemedtsson et al., 1997).

4.2.1 Direct measurement of the NEE by eddy covariance

The accuracy of the EC method depends upon a number of random and systematic errors related to sampling and theoretical issues as well as to site specific matters such as limited fetch or topographic constraints. However, with a careful system design and the correct field installation, random errors are generally small (2–3 %) and derive from errors related to gas analyzer calibration, signal dampening in tubes, and sensor separation. Errors in EC measurements are generally considered to be less than 7 % during daytime and less than 12 % during nighttime conditions but for longer time periods they typically decrease to ± 5 % due to the increasing number of samples (Baldocchi, 2003). In addition to random error, uncertainties due to gap filling, limited fetch conditions and potential impacts of advection flows need to be considered. Advection occurs mainly during stable nighttime conditions, periods we have filtered out by the u^* threshold method, which has been shown to be the best available approach (Aubinet et al., 2010). The gap filling that we undertook means that the modeled flux rates might introduce a bias. Moffat et al. (2007) estimated this error to be in the order of $\pm 25\,\mathrm{g\,C\,m^{-2}\,yr^{-1}}$, which would add a potential error of $0.25\,\mathrm{t\,C\,m^{-2}}$ to our direct estimate of $\mathrm{NEE_{meas}}$. As a conservative method to estimate the uncertainty of the NEE we used an error of ±15 % (Fig. 3), which gives a maximum uncertainty range between −2.3 and $-1.7\,\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$.

Errors related to limitations in the fetch and topographical constraints are more difficult to quantify. In Sweden, afforested organic soils are often located in small valleys within a mosaic landscape, which means they are not ideal for applying micrometeorological techniques. This does apply to the Skogaryd research catchment but, due to the prevailing wind being from the SW, the fetch extends mainly over organic soils with uniform stands (Fig. 1b). However, we cannot exclude the possibility that the $\mathrm{NEE_{meas}}$ may have been affected by fluxes from mineral soils that are likely to be lower than fluxes from organic soils due to their lower C content. This would have resulted in an overestimation of the CO_2 uptake. In order to quantify this effect, long-term series are needed to examine the relationship between the NEE and the wind direction.

4.2.2 Indirect quantification of the NEE

The magnitude of $\mathrm{NEE_{calc}}$ in approach 2 is heavily based on precise quantification of the respiration from SOM (R_{SOM}). Generally, the total heterotrophic soil efflux R_{h} is one of the largest fluxes in ecosystems (Hanson et al., 2000; Schlesinger et al., 2000), which implies that even small uncertainties in the components of this flux (see Eq. 5) can equate to the size of the total net C flux (Ojanen et al., 2012).

Due to the use of a trenching experiment, the measured soil CO_2 efflux ($E_{\mathrm{CO_2}}$) had to be corrected for the decay of roots and mycelia as well as for the CO_2 flux from the litter layer (Table 1). This correction accounts for 38 % of the measured efflux. Because of the component fluxes of R_{SOM}, the CO_2 flux from the litter layer, R_{LL} (Eq. 10a) has a high uncertainty that derives from the high variation in aboveground litter fall ($2.9 \pm 1.1\,\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$). Uncertainties in the calculation of the annual C accumulation in the litter layer only had a minor impact on the total standard deviation of R_{SOM}.

Assuming an uncertainty in the measured dry weight of 20 % resulted in a R_{LL} in the range 2.7 (± 1.1) to 2.77 (± 1.1) $\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$. An alternative method applied previously is to remove the litter layer before the experiment and install a mesh to exclude fresh litter (Mäkiranta et al., 2007). However, this method generates uncertainty in quantifying the amount of C removed and, in addition, causes artifacts due to changes in soil surface conditions, such as temperature, humidity and, consequently, priming effects (Kuzyakov, 2002; Mäkiranta et al., 2012).

Uncertainties in the correction terms for root decay were smaller and derive from potential errors in the applied root decomposition rates (see Table 1). Root decomposition is closely related to root turnover, which has been shown to be highly variable depending on tree species and also on site-specific and climatic conditions (Finer et al., 2011; Brunner et al., 2012). We assessed this uncertainty by calculating the root decay for different decomposition rates as given by Ngao et al. (2007) and Chen et al. (2001). An alternative approach to estimate the impact of trenched roots and mycelia was applied by Comstedt et al. (2011). Using a modeling approach based on a trenching experiment, they quantified the overestimation of the measured soil efflux due to decaying roots and mycelia to be 16 % in the first year. However, their model uses the CO_2 flux in the second year after trenching as a reference, assuming that the direct "decomposition effect" can be discounted that early. This is in sharp contrast to the results of Mäkiranta et al. (2010), who showed that the impact of decaying roots and mycelia can be considerable even in the third and fourth year after trenching. The estimation is, therefore, probably too low. Comstedt et al. (2011) furthermore pointed to the impact of trenching on the soil water level, which rises due to reduced root uptake so that microbial activity in the trenched plots may decrease. They concluded that this effect leads to an overestimation of the measured soil CO_2 efflux of 29 %. In this study we did not observe any effect of trenching on the water table depth, so this effect will have been minor.

One further concern with regard to the application of the chamber technique is that the technique itself creates artifacts that are difficult to estimate (Livingston and

Hutchinson, 1995). Pumpanen et al. (2004) showed that, depending on the chamber design, measurements of the same source by several chambers can differ by as much as 35 %. In particular, the use of a chamber without ventilation seems to underestimate fluxes (Christiansen et al., 2011). These possible artifacts caused by the use of the chamber method itself naturally also apply to the measurement of CH_4 and N_2O fluxes, which were determined using manual chambers in the current study. In contrast, a recent publication by Wang et al. (2012) demonstrated that CH_4 fluxes as determined by the chamber method (without ventilation) and the eddy covariance were very similar.

4.3 Losses due to discharge

The export of C from the soil to streams and ditches is a factor with potential impact on the ecosystem C budget (Nilsson et al., 2008) that was not quantified in the present study. For afforested organic soils, total organic C (TOC) export ranges between 58 and 115 $\mathrm{kg\,C\,ha^{-1}\,yr^{-1}}$ and depends on a variety of factors including the organic matter content, discharge during the spring melting period as well as the magnitude of winter precipitation (Laudon et al., 2004; Köhler et al., 2008; Haei et al., 2010; Rantakari et al., 2010; Dyson et al., 2011). Due to the high C content and the high winter precipitation in 2008 the TOC export in Skogaryd was probably higher than that reported in studies by Köhler et al. (2008) and Dyson et al. (2011) where soils had a lower C content. However, there was no consistent snow layer in Skogaryd in 2008, so that the impact of the spring melting period was probably negligible. Total inorganic C (TIC) export reported in the literature seems to be of minor importance ($< 30\,\mathrm{kg\,C\,ha^{-1}\,yr^{-1}}$). Overall, with reported maxima of 30 and 115 $\mathrm{kg\,C\,ha^{-1}\,yr^{-1}}$, the contribution of discharge C losses to the total C budget at Skogaryd is probably low (less than 6 % of the net C flux).

4.4 Total greenhouse gas budget for the Skogaryd research catchment

The measured CH_4 flux was very low and always negative, i.e. CH_4 was taken up by the soil. These results ~~correspond~~ to values previously reported for drained organic soils in Sweden, Finland, the UK and Germany for different types of coniferous forests (-0.18 and $-2.0\,\mathrm{kg\,C\,ha^{-1}\,yr^{-1}}$)~~, which are rather consistent between sites~~ (e.g. Kasimir Klemedtsson et al., 1997; Minkkinen et al., 2002; Huttunen et al., 2003; Maljanen et al., 2003a; von Arnold et al., 2005; Yamulik et al., 2012). Due to the low magnitude of the flux, the contribution to the GHG budget is negligible (Table 4). Although CH_4 fluxes were always found to be negative, the source potential at the site might be underestimated due to possible CH_4 emissions from ditches, which were not considered in this study. Previous studies have shown that, depending on the water movement and water temperature, CH_4 emissions can be 10 times as high as from neighboring soil (von Arnold et al., 2005; Minkkinen and Laine, 2006).

In contrast to CH_4, significant N_2O fluxes of 11.0 (± 8.0) $\mathrm{kg\,N_2O\,ha^{-1}\,yr^{-1}}$ were observed at the upper end of the reported range of values (0.4–8.1 $\mathrm{kg\,N_2O\,ha^{-1}\,yr^{-1}}$: Maljanen et al., 2003b; von Arnold et al., 2005; Alm et al., 2007; Yamulik et al., 2012) confirming that N_2O emissions are highest from soils which were either rather nutrient-rich (Alm et al., 2007) or in agricultural use previously (Maljanen et al., 2003b). In a review by Alm et al. (2007), N_2O effluxes of up to 10 $\mathrm{kg\,N_2O\,ha^{-1}\,yr^{-1}}$ are reported for nutrient-rich well-drained pine fens and spruce mires.

The comparatively high N_2O emissions from the present site can partly be explained by the soil C : N ratio of 22 to 23. Klemedtsson et al. (2005) found that N_2O emissions rapidly increase with reductions in the soil C : N below a threshold value of 25. Additionally, the water table level has been identified as one of the determining factors for N_2O emissions (Martikainen et al., 1993). The well-drained organic soil at the Skogaryd research catchment has a water table of below 40 cm, which is a relatively low level compared to the sites mentioned above; this, together with the high soil nutrient content, explains the comparatively high N_2O emissions.

Considering all GHG fluxes, the site is either a GHG sink or source, depending on the approach applied to determine the site C budget (Table 4). Our results contrast with the findings of Ojanen et al. (2012) who compared the EC technique and the chamber technique + balance calculations to determine soil C balance. They found only marginal differences between the respective soil effluxes. However, they pointed out that the chamber estimation depends largely on assumed turnover rates of fine roots to calculate belowground litter production. Using the lowest values of their given range, the two approaches differed by almost 2 t C ha^{-1} yr^{-1}. Furthermore they did not, to our knowledge, correct for the decay of roots in trenched measurement collars nor did they consider the impact of mycorrhizal mycelia. These factors can lead to either an underestimation (e.g. due to missing mycelia litter input) or an overestimation of the soil C losses (e.g. due to an overestimation of the measured heterotrophic flux) so that it is difficult to estimate the net effect.

As demonstrated above, approach 2 is particularly susceptible to errors due to the high number of component fluxes and uncertainties involved in their determination. This applies primarily to measurement of aboveground litter fall and the calculation of the decay of trenched roots, which is very sensitive to the choice of root decomposition rates (see Table 1). In conclusion, we consider approach 1, the EC technique, to be the more reliable tool to obtain precise estimates of the ecosystem C budget. This approach indicates that the study site is a GHG sink of -4.1 t CO_{2eq} ha^{-1} yr^{-1} and the N_2O efflux offsets about 50 % of the C sink strength of the site (Table 4).

It must be stressed that we only present measurements for a single year, so that no evidence can be provided to elucidate the long-term GHG budget of this site. A few studies, however, give some indication of the long-term patterns. Using snap-shot measurements and modeling, Hargreaves et al. (2003) showed that young forests of about 4 to 8 yr act as a net C source but then ~~transform into~~ C sinks when the C ~~accumulation in~~ trees exceeds the soil C losses due to decomposition. With increasing age, i.e. after canopy closure, the C sink potential declines again because of the decomposition of harvest residues and the lower C accumulation by trees (Ågren and Hyvönen, 2003;

Black and Gallagher, 2010). As a consequence, the overall net GHG uptake of the investigated site in the Skogaryd research catchment is likely to be limited to a rather short time when the forest is at its maximum productivity.

~~Afforested nutrient-rich organic soils might therefore be net GHG emitters for a rather long period.~~ Emissions of CO_2 and N_2O can be high, as has been shown in previous studies (e.g. von Arnold et al., 2005; Alm et al., 2007) and can exceed the C accumulation in trees. The afforestation of former agricultural soils in order to turn the ecosystem into an overall long-term GHG sink might therefore not be an effective strategy. The drainage level has been shown to have a considerable impact on GHG emissions (Maljanen et al., 2003a), e.g. in a recent study by Yamulki et al. (2012), forests on undrained organic soils had much lower soil CO_2 effluxes than drained forest sites. The regulation of the water table by reducing the cleaning of ditches, for example, might be a suitable tool for reducing GHG emissions.

5 Conclusions

We investigated the total GHG budget of a spruce forest on a nutrient-rich organic soil using two approaches: an indirect budget calculation based on chamber measurements and a direct measurement of the NEE by eddy covariance. Due to major uncertainties involved in the indirect method, the direct method was considered to be the more reliable tool for obtaining precise estimates of C and the overall GHG budget. According to the EC method, the site is currently a GHG sink due to high C uptake by the actively growing forest; this is, however, offset to a large extent by N_2O emissions. We conclude that during peak forest production, the nutrient rich forest ecosystem is a GHG sink, but it will probably be a GHG source over the remaining time, hence contributing to global warming. According to Klemedtsson et al. (2005) and Ernfors et al. (2008), organic soils with a low C : N ratio (< 25), as is typical for former agricultural soils, cover about 60 % of the total area of drained organic forestland in Sweden, so that the GHG balance of these sites is of major importance for the national GHG inventory. The long-term GHG sink potential of these sites must, therefore, be seriously questioned and it is recommended that future GHG mitigation strategies take site-specific properties into account.

Acknowledgements. Financial support came from the Swedish Research Council for Environment, Agricultural Sciences and Spatial Planning (FORMAS), the strategic research area BECC (Biodiversity and Ecosystem services in a Changing Climate, http://www.cec.lu.se/research/becc), the Linnaeus Centre LUCCI (Lund University Centre for studies of Carbon Cycle and Climate Ineractions) as well as the NitroEurope IP under the EC 6th Framework Programme (Contract No. 017841). We sincerely thank Johannes Edvardsson, Lund University, for providing the data on tree age. We also thank David Allbrand for his assistance in the field.

References

Ågren, G. I. and Hyvönen, R.: Changes in carbon stores in Swedish forest soils due to increased biomass harvest and increased temperatures analysed with a semi-empirical model, Forest Ecol. Manag., 174, 25–37, 2003.

Alm, J., Shurpali, N., Minkkinen, K., Aro, L., Hytönen, J., Laurila, T., Lohila, A., Maljanen, M., Martikainen, P., Mäkiranta, P., Penttila, T., Saarnio, S., Silvan, N., Tuittila, E., and Laine, J.: Emission factors and their uncertainty for the exchange of CO_2, CH_4 and N_2O in Finnish managed peatlands, Boreal Environ. Res., 12, 191–209, 2007.

Aubinet, M., Feigenwinter, C., Heinesch, B., Bernhofer, C., Canepa, E., Lindroth, A., Montagnani, L., Rebmann, C., Sedlak, P., and Van Gorsel, E.: Direct advection measurements do not help to solve the night-time CO_2 closure problem: evidence from three different forests, Agr. Forest Meteorol., 150, 655–664, 2010.

Baldocchi, D. D.: Assessing the eddy covariance technique for evaluating carbon dioxide exchange rates of ecosystems: past, present and future, Global Change Biol., 9, 479–492, 2003.

Berg, B. and C. McClaugherty (Eds.): Plant Litter: Decomposition, Humus Formation, Carbon Sequestration, 2nd edition, Springer, Berlin, Germany, 2008.

Berglund, O. and Berglund, K.: Distribution and cultivation intensity of agricultural peat and gyttja soils in Sweden and estimation of greenhouse gas emissions from cultivated peat soils, Geoderma, 154, 173–180, 2010.

Black, K. and Gallagher, G.: The greenhouse gas balance of peatland forest, COFORD Connects, Forestry and Environment, 11, Dublin, Ireland, 2010.

Brunner, I., Bakker, M., Björk, R., Hirano, Y., Lukac, M., Aranda, X., Børja, I., Eldhuset, T., Helmisaari, H., Jourdan, C., Konopka, B., Lopez, B., Miguel Perez, C., Persson, H., and Ostonen, I.: Fine-root turnover rates of European forests revisited: an analysis of data from sequential coring and ingrowth cores, Plant Soil, 362, 357–372, 2013.

Chen, H., Harmon, M. E., and Griffiths, R. P.: Decomposition and nitrogen release from decomposing woody roots in coniferous forests of the Pacific Northwest: a chronosequence approach, Can. J. Forest Res., 31, 246–260, 2001.

Christiansen, J. R., Korhonen, J. F. J., Juszczak, R., Giebels, M., and Pihlatie, M.: Assessing the effects of chamber placement, manual sampling and headspace mixing on CH_4 fluxes in a laboratory experiment, Plant Soil, 343, 171–185, 2011.

Comstedt, D., Boström, B., and Ekblad, A.: Autotrophic and heterotrophic soil respiration in a Norway spruce forest: estimating the root decomposition and soil moisture effects in a trenching experiment, Biogeochemistry, 104, 121–132, 2011.

Couwenberg, J.: Greenhouse gas emission from managed peat soils: is the IPCC resporting realistic?, Mires Peat, 8, 1–10, 2011.

Davidson, E. A. and Janssens, I. A.: Temperature sensitivity of soil carbon decomposition and feedbacks to climate change, Nature, 440, 165–173, 2006.

Dunn, A., Barford, C., Wofsy, S., Goulden, M., and Daube, B.: A long-term record of carbon exchange in a boreal black spruce forest: means, responses to interannual variability, and decadal trends, Global Change Biol., 13, 577–590, 2007.

Dyson, K. E., Billett, M. F., Dinsmore, K. J., Harvey, F., Thomson, A. M., Piirainen, S., and Kortelainen, P.: Release of aquatic carbon from two peatland catchments in E. Finland during the spring snowmelt period, Biogeochemistry, 103, 125–142, 2011.

Ernfors, M.: Greenhouse gas fluxes between drained forested peatlands and the atmosphere – influence of nutrient status and wood ash fertilization. PhD thesis, University of Gothenburg, Gothenburg, Sweden, 2009.

Ernfors, M., von Arnold, K., Stendahl, J., Olsson, M., and Klemedtsson, L.: Nitrous oxide emissions from drained organic forest soils – an up-scaling based on C : N ratios, Biogeochemistry, 89, 29–41, 2008.

Ernfors, M., Rütting, T., and Klemedtsson, L.: Increased nitrous oxide emissions from a drained organic forest soil after exclusion of ectomycorrhizal mycelia, Plant Soil, 343, 161–170, 2011.

Finer, L., Ohashi, M., Noguchi, K., and Hirano, Y.: Fine root production and turnover in forest ecosystems in relation to stand and environmental characteristics, Forest Ecol. Manag., 262, 2008–2023, 2011.

Flanagan, L. B. and Syed, K. H.: Stimulation of both photosynthesis and respiration in response to warmer and drier conditions in a boreal peatland ecosystem, Global Change Biol., 17, 2271–2287, 2011.

Forster, P., Ramaswamy, V., Artaxo, P., Berntsen, T., Betts, R., Fahey, D., Haywood, J., Lean, J., Lowe, D., Myhre, G., Nganga, J., Prinn, R., Raga, G., Schulz, M., and van Dorland, R.: Changes in atmospheric constituents and in radiative forcing, in: Climate Change 2007: The Physical Science Basis, Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, edited by: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K., Tignor, M., and Miller, H., Cambridge University Press, Cambridge, UK, 133–234, 2007.

Grelle, A. and Lindroth, A.: Eddy-correlation system for long-term monitoring of fluxes of heat, water vapour and CO_2, Global Change Biol., 2, 297–307, 1996.

Haei, M., Oquist, M. G., Buffam, I., Ågren, A., Blomkvist, P., Bishop, K., Lofvenius, M. O., and Laudon, H.: Cold winter soils enhance dissolved organic carbon concentrations in soil and stream water, Geophys. Res. Lett., 37, L08501, doi:10.1029/2010GL042821, 2010.

Hanson, P. J., Edwards, N. T., Garten, C. T., and Andrews, J. A.: Separating root and soil microbial contributions to soil respiration: a review of methods and observations, Biogeochemistry, 48, 115–146, 2000.

Hargreaves, K., Milne, R., and Cannell, M.: Carbon balance of afforested peatland in Scotland, Forestry, 76, 299–317, 2003.

Höper, H.: Carbon and nitrogen mineralization rates in German agriculturally used fenlands, in: Wetlands in Central Europe: Soil Organisms, Soil Ecological Processes and Trace Gas Emissions, edited by: Broll, G., Merbach, W., and Pfeiffer, E. M., Springer, Berlin, Germany, 149–164, 2002.

Huttunen, J., Nykanen, H., Martikainen, P., and Nieminen, M.: Fluxes of nitrous oxide and methane from drained peatlands following forest clear-felling in southern Finland, Plant Soil, 255, 457–462, 2003.

Jalkanen, A., Mäkipää, G., Ståhl, G., Lehtonen, A., and Petersson, H.: Estimation of the biomass stock of trees in Sweden: comparison of biomass equations and age-dependent biomass expansion factors, Ann. For. Sci., 62, 845–851, 2005.

Jenkins, J. C., Chojnacky, D. C., Heath, L. S., and Birdsey, R. A.: National-scale biomass estimators for United States tree species, Forest. Sci., 49, 12–35, 2003.

Kasimir-Klemedtsson, Å., Klemedtsson, L., Berglund, K., Martikainen, P., Silvola, J., and Oenema, O.: Greenhouse gas emissions from farmed organic soils: a review, Soil Use Manage., 13, 245–250, 1997.

Kasimir Klemedtsson, Å., Weslien, P., and Klemedtsson, L.: Methane and nitrous oxide fluxes from a farmed Swedish Histosol, Eur. J. Soil Sci., 60, 321–331, 2009.

Klemedtsson, L., von Arnold, K., Weslien, P., and Gundersen, P.: Soil CN ratio as a scalar parameter to predict nitrous oxide emissions, Global Change Biol., 11, 1142–1147, 2005.

Klemedtsson, L., Ernfors, M., Björk, R., Weslien, P., Rütting, T., Crill, P., and Sikström, U.: 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, 2010.

Kljun, N., Black, T., Griffis, T., Barr, A., Gaumont-Guay, D., Morgenstern, K., McCaughey, J., and Nesic, Z.: Response of net ecosystem productivity of three boreal forest stands to drought, Ecosystems, 9, 1128–1144, 2006.

Köhler, S. J., Buffam, I., Laudon, H., and Bishop, K. H.: Climates control of intra-annual and interannual variability of total organic carbon concentration and flux in two contrasting boreal landscape elements, J. Geophys. Res.-Biogeo., 113, G03012, doi:10.1029/2007JG000629, 2008.

Kormann, R. and Meixner, F. X.: An analytical footprint model for non-neutral stratification, Bound.-Lay. Meteorol., 99, 207–224, 2001.

Kuzyakov, Y.: Review: factors affecting rhizosphere priming effects, J. Plant Nutr. Soil Sc., 165, 382–396, 2002.

Laiho, R. and Finer, L.: Changes in root biomass after water-level drawdown on pine mires in southern Finland, Scand. J. Forest Res., 11, 251–260, 1996.

Laudon, H., Köhler, S., and Buffam, I.: Seasonal TOC export from seven boreal catchments in northern Sweden, Aquat. Sci., 66, 223–230, 2004.

Lindroth, A., Klemedtsson, L., Grelle, A., Weslien, P., and Langvall, O.: Measurement of net ecosystem exchange, productivity and respiration in three spruce forests in Sweden shows unexpectedly large soil carbon losses, Biogeochemistry, 89, 43–60, 2008.

Livingston, G. P. and Hutchingson, G. L.: Enclosure-based measurement of trace gas exchange: application and sources of error, in: Methods in Ecology-Biogenic Trace Gases:

Measuring Emissions from Soil and Water, edited by: Matson, P. A. and Harriss, R. C., Blackwell, London, UK, 14–51, 1995.

Lohila, A., Laurila, T., Aro, L., Aurela, M., Tuovinen, J. P., Laine, J., Kolari, P., and Minkkinen, K.: Carbon dioxide exchange above a 30-year-old Scots pine plantation established on organic-soil cropland, Boreal Environ. Res., 12, 141–157, 2007.

Lohila, A., Minkkinen, K., Aurela, M., Tuovinen, J.-P., Penttilä, T., Ojanen, P., and Laurila, T.: Greenhouse gas flux measurements in a forestry-drained peatland indicate a large carbon sink, Biogeosciences, 8, 3203–3218, doi:10.5194/bg-8-3203-2011, 2011.

Mäkiranta, P., Hytönen, J., Aro, L., Maljanen, M., Pihlatie, M., Potila, H., Shurpali, N., Laine, J., Lohila, A., Martikainen, P., and Minkkinen, K.: Soil greenhouse gas emissions from afforested organic soil croplands and cutaway peatlands, Boreal Environ. Res., 12, 159–175, 2007.

Mäkiranta, P., Riutta, T., Penttila, T., and Minkkinen, K.: Dynamics of net ecosystem CO_2 exchange and heterotrophic soil respiration following clearfelling in a drained peatland forest, Agr. Forest Meteorol., 150, 1585–1596, 2010.

Mäkiranta, P., Laiho, R., Penttilä, T., and Minkkinen, K.: The impact of logging residue on soil GHG fluxes in a drained peatland forest, Soil Biol. Biochem., 48, 1–9, 2012.

Maljanen, M., Liikanen, A., Silvola, J., and Martikainen, P.: Methane fluxes on agricultural and forested boreal organic soils, Soil Use Manage., 19, 73–79, 2003a.

Maljanen, M., Liikanen, A., Silvola, J., and Martikainen, P.: Nitrous oxide emissions from boreal organic soil under different land-use, Soil Biol. Biochem., 35, 689–700, 2003b.

Maljanen, M., Komulainen, V. M., Hytönen, J., Martikainen, P., and Laine, J.: Carbon dioxide, nitrous oxide and methane dynamics in boreal organic agricultural soils with different soil characteristics, Soil Biol. Biochem., 36, 1801–1808, 2004.

Maljanen, M., Hytönen, J., Mäkiranta, P., Alm, J., Minkkinen, K., Laine, J., and Martikainen, P. J.: Greenhouse gas emissions from cultivated and abandoned organic croplands in Finland, Boreal Environ. Res., 12, 133–140, 2007.

Maljanen, M., Sigurdsson, B. D., Gudmundsson, J., Óskarsson, H., Huttunen, J. T., and Martikainen, P. J.: Greenhouse gas balances of managed peatlands in the Nordic countries – present knowledge and gaps, Biogeosciences, 7, 2711–2738, doi:10.5194/bg-7-2711-2010, 2010.

Marklund, L.: Biomassfunktioner för tall, gran och björk i Sverige, Rapporter Skog, 45, Sveriges Lantbruksuniversitet, Sweden, 1–73, 1988.

Martikainen, P., Nykänen, H., Crill, P., and Silvola, J.: Effect of a lowered water-table on nitrous-oxide Fluxes from Northern peatlands, Nature, 366, 51–53, 1993

Mehtätalo, L.: A longitudinal height-diameter model for Norway spruce in Finland, Can. J. Forest Res., 34, 131–140, 2004.

Minkkinen, K. and Laine, J.: Vegetation heterogeneity and ditches create spatial variability in methane fluxes from peatlands drained for forestry, Plant Soil, 285, 289–304, 2006.

Minkkinen, K., Vasander, H., Jauhiainen, S., Karsisto, M., and Laine, J.: Post-drainage changes in vegetation composition and carbon balance in Lakkasuo mire, Central Finland, Plant Soil, 207, 107–120, 1999.

Minkkinen, K., Laine, J., and Hokka, H.: Tree stand development and carbon sequestration in drained peatland stands in Finland – a simulation study, Silva Fenn, 35, 55–69, 2001.

Minkkinen, K., Korhonen, R., Savolainen, I., and Laine, J.: Carbon balance and radiative forcing of Finnish peatlands 1900–2100 – the impact of forestry drainage, Global Change Biol., 8, 785–799, 2002.

Moffat, A. M., Papale, D., Reichstein, M., Hollinger, D. Y., Richardson, A. D., Barr, A. G., Beckstein, C., Braswel, B. H., Churkina, G., Desai, A. R., Falge, E., Gove, J. H., Heimann, M., Hui, D., Jarvis, A. J., Kattge, J., Noormets, A., and Stauch, V. J.: Comprehensive comparison of gap-filling techniques for eddy covariance net carbon fluxes, Agr. Forest Meteorol., 147, 209–232, 2007.

Ngao, J., Longdoz, B., Granier, A., and Epron, D.: Estimation of autotrophic and heterotrophic components of soil respiration by trenching is sensitive to corrections for root decomposition and changes in soil water content, Plant Soil, 301, 99–110, 2007.

Nilsson, M., Sagerfors, J., Buffam, I., Laudon, H., Eriksson, T., Grelle, A., Klemedtsson, L., Weslien, P., and Lindroth, A.: Contemporary carbon accumulation in a boreal oligotrophic minerogenic mire – a significant sink after accounting for all C-fluxes, Global Change Biol., 14, 2317–2332, 2008.

Nykänen, H., Alm, J., Silvola, J., Tolonen, K., and Martikainen, P.: Methane fluxes on boreal peatlands of different fertility and the effect of long-term experimental lowering of the water table on flux rates, Global Biogeochem. Cy., 12, 53–69, 1998.

Ojanen, P., Minkkinen, K., Alm, J., and Penttila, T.: Soil-atmosphere CO_2, CH_4 and N_2O fluxes in boreal forestry-drained peatlands, Forest Ecol. Manag., 260, 411–421, 2010.

Ojanen, P., Minkkinen, K., Lohila, A., Badorek, T., and Penttila, T.: Chamber measured soil respiration: a useful tool for estimating the carbon balance of peatland forest soils?, Forest Ecol. Manag., 277, 132–140, 2012.

Pihlatie, M. K., Kiese, R., Brüggemann, N., Butterbach-Bahl, K., Kieloaho, A.-J., Laurila, T., Lohila, A., Mammarella, I., Minkkinen, K., Penttilä, T., Schönborn, J., and Vesala, T.: Greenhouse gas fluxes in a drained peatland forest during spring frost-thaw event, Biogeosciences, 7, 1715–1727, doi:10.5194/bg-7-1715-2010, 2010.

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., Østreng, G., Ziegler, W., Anthoni, P., Lindroth, A., and Hari, P.: Comparison of different chamber techniques for measuring soil CO_2 efflux, Agr. Forest Meteorol., 123, 159–176, 2004.

Rantakari, M., Mattsson, T., Kortelainen, P., Piirainen, S., Finer, L., and Ahtiainen, M.: Organic and inorganic carbon concentrations and fluxes from managed and unmanaged boreal first-order catchments, Sci. Total Environ., 408, 1649–1658, 2010.

Schlesinger, W. H. and Andrews, J. A.: Soil respiration and the global carbon cycle, Biogeochemistry, 48, 7–20, 2000.

Smith, K. A. and Conen, F.: Impacts of land management on fluxes of trace greenhouse gases, Soil Use Manage., 20, 255–263, 2004.

Vickers, D., Irvine, J., Martin, J. G., and Law, B. E.: Nocturnal subcanopy flow regimes and missing carbon dioxide, Agr. Forest Meteorol., 152, 101–108, 2012.

von Arnold, K.: Forests and Greenhouse Gases – Fluxes of CO_2, CH_4 and N_2O from Drained Forests on Organic Soils, PhD thesis, Linköping Studies in Arts and Science, 302, Linköping University, Sweden, 2004.

von Arnold, K., Weslien, P., Nilsson, M., Svensson, B., and Klemedtsson, L.: Fluxes of CO_2, CH_4 and N_2O from drained coniferous forests on organic soils, Forest Ecol. Manag., 210, 239–254, 2005.

Wallander, H., Ekblad, A., Godbold, D., Johnson, D., Bahr, A., Baldrian, P., Björk, R., Kieliszewska-Rokicka, B., Kjøller, R., Kraigher, H., Plassard, C., and Rudawska, M.: Evaluation of methods to estimate production, biomass and turnover of ectomycorrhizal mycelium in forests soils – a review, Soil Biol. Biochem., 57, 1034–1047, 2013.

Wallin, G., Linder, S., Lindroth, A., Rantfors, M., Flemberg, S., and Grelle, A.: Carbon dioxide exchange in Norway spruce at the shoot, tree and ecosystem scale, Tree Physiol., 21, 969–976, 2001.

Wang, J. M., Murphy, J. G., Geddes, J. A., Winsborough, C. L., Basiliko, N., and Thomas, S. C.: Methane fluxes measured by eddy covariance and static chamber techniques at a temperate forest in central Ontario, Canada, Biogeosciences Discuss., 9, 17743–17774, doi:10.5194/bgd-9-17743-2012, 2012.

Watson, R. T., Noble, I. R., Bolin, B., Ravidranath, N. H., Verardo, D. J., and Dokken, D. J.: Land Use, Land Use Change, and Forestry. A special report of the IPCC, Cambridge University Press, Cambridge, UK, 2000.

Yamulki, S., Anderson, R., Peace, A., and Morison, J. I. L.: Soil CO_2, CH_4 and N_2O fluxes from an afforested lowland raised peatbog in Scotland: implications for drainage and restoration, Biogeosciences, 10, 1051–1065, doi:10.5194/bg-10-1051-2013, 2013.

Table 1. Biomass in autumn 2007 ($\mathrm{t\,C\,ha^{-1}}$) and respective parameter values of fine roots, coarse roots, and fungal mycelia used for the calculation of the CO_2 flux from decaying roots and mycelia (R_{decay}). If not stated otherwise, parameter values were used as given in Ngao et al. (2007). α = microbial efficiency, k_r = decomposition constant.

	Biomass ($\mathrm{t\,C\,ha^{-1}}$)	Determined by method	α	k_r ($\mathrm{yr^{-1}}$)	Calculated flux (± stand. dev.) ($\mathrm{t\,C\,ha^{-1}}$)
Fine roots (< 1 cm)	6.2	measured[a]	0.4[b]	0.5	1.5 (0.4)[c]
Coarse roots	23.4	Minkkinen et al. (2001)	0.3	0.045[d]	0.7 (0.5)[c]
Fungal mycelia	0.023	measured[a]	0.4[b]	0.5	< 0.00

[a] By R. G. Björk (personal communication, 2011).

[b] Microbial efficiency was assumed to be higher for fine roots and fungal mycelia compared to coarse roots.

[c] Standard deviation was determined as the uncertainty in the calculated flux for a decomposition rate k_r of ± 50 %.

[d] k_r was adapted for coarse roots and determined acc. to Chen et al. (2001).

Table 2. CH_4 (kg CH_4 ha^{-1} yr^{-1}) and N_2O (kg N_2O ha^{-1} yr^{-1}) fluxes and the corresponding flux in CO_2 equivalents (t CO_{2eq} ha^{-1} yr^{-1}) (± stand. dev.) at the Skogaryd research catchment for 2008. CO_2 equivalents were calculated based for a 100 yr time frame based on their respectively global warming potential (IPCC: Forster et al., 2007).

	station 1	station 2	station 3	mean flux	CO_2 equivalents (t CO_{2equ} ha^{-1} yr^{-1})
N_2O flux (kg N_2O ha^{-1} yr^{-1})	3.1	11.1	18.7	11.0 (± 8.0)	3.28 (± 2.4)
CH_4 flux (kg CH_4 ha^{-1} yr^{-1})	−5.8	−1.7	−5.5	−4.4 (± 2.2)	−0.11 (± 0.05)

Table 3. Annual tree growth, NEE and respiration from SOM determined at comparable organic sites. If not stated otherwise, units are $\mathrm{t\,C\,ha^{-1}\,yr^{-1}}$.

Reference	Site	Period	Species	Forest age (yr)	Stem density (trees $\mathrm{ha^{-1}}$)	Tree growth	NEE	SOM respiration
von Arnold et al. (2005)	Sweden, Asa	2000–02	Norway spruce	90	750	3.5		
Lohila et al. (2011)	Finland, Kalevansuo	2005	natural pine and birch		835	1.6	−2.4	
Lohila et al. (2004)	Finland, Allkia	2003	Scots pine	ca. 30	977	2.4	0.5	−0.8
Mäkiranta et al. (2007)	Finland, Alkkia	2003–04	Scots pine	32				4.8
Mäkiranta et al. (2010)	Finland, Padasjoki	2001–04	Scots pine	60		1.9		4.1
Ojanen et al. (2010)	Finland, 68 sites	2007–08	coniferous					1.5 to 6.7
This study	*Sweden, Skogaryd*	*2008*	*Spruce dominated*	*60*	*985*	*8.2*	*0.0^{b} to -2.0^{a}*	*6.2^{a} to 8.2^{b}*

[a] Approach 1: EC.

[b] Approach 2: chambers.

Table 4. Total GHG budget at the Skogaryd Research Catchment, presented as CO_2 equivalents determined by the measured N_2O and CH_4 flux and two different approaches to determining the NEE. All units are $t\,CO_{2equ}\,ha^{-1}\,yr^{-1}$.

		Approach 1 (GHG flux ± stdev)		Approach 2 (GHG flux ± stdev)	
plant CO_2 flux					
tree growth	ΔB	−30.0	6.2	−30.0	6.2
soil CO_2 flux					
measured soil flux	E_{CO_2}			48.0	6.9
respiration from litter layer	R_l			9.9	3.9
decay from root, mycelia	R_{decay}			8.1	1.3
soil net efflux	R_{res}/R_{SOM}	22.7		30.0	8.1
ecosystem CO_2 flux	NEE	−7.3	−1.1	0.0	10.2
other GHG					
CH_4 flux		0.0	0.0	0.0	0.0
N_2O flux		3.3	2.4	3.3	2.4
total GHG flux		−4.1	2.6	3.3	10.1

Table A1. Overview of the instrumentation applied at the Skogaryd research catchment: measured environmental parameter or technical equipment, height of the instrument at the eddy flux tower or soil depth of the instrument, information on applied software or instrument, producing company.

Parameter/technical equipment	Instrument height/depth (m)	Instrument	Producer
abiotic data: recorded every 30 min			
air temperature	1.5	Campbell 107 Temperature Probe	Campbell Scientific, Logan UT, USA
radiation shield		UT12VA 12-Plate Radiation Shield	Campbell Scientific, Logan UT, USA
soil moisture	−0.05 to −0.6	CS616 Water Content reflectometers	Campbell Scientific, Logan UT, USA
soil temperature	−0.05 to −0.6	copper-constantan thermocouples, Temperature probes 107	Campbell Scientific, Logan UT, USA
logger system for soil temperature and moisture record		CR10X logger, AM16/32A multiplexer	Campbell Scientific, Logan UT, USA
NEE: measured by eddy covariance technique			
air temperature, air humidity	22	MP101A probe	Rotronic, USA
wind speed	27	sonic anemometer Gill R3	Solent, UK
CO_2, H_2O concentration		infrared gas analyser LI-6262	Li-Cor Inc., Lincoln USA
global radiation	27	pyranometer SP-series	Apogee SP, Logan, USA
net radiation	27	net radiometer NR-Lite	Kipp & Zonen, Delft, Netherlands
PAR	27	PAR sensor SQ-series	Apogee SP, Logan, USA
Soil respiration: automatic closed chambers (0.24 m^3), ventilated by fan, automatic lid closure before measurement			
Actuator for lid opening		Linear Actuator LA12	LINAK A/S, Nordborg, Denmark
logger system		CR10X logger, AM16/32A multiplexer, SDM-CD16 relay control port module	Campbell Scientific, Logan UT, USA
Infrared gas analyser		SBA-4 OEM CO2 Analyzer	PP Systems, Amesbury MA, USA
N_2O and CH_4 flux: dark chamber technique, gas sampling 4, 8, 16, and 32 min after chamber closure by means of extracting a gas sample with a syringe inserted into a rubber septum in the chamber lid.			
determination of gas concentration of samples		Varian 3400 gas chromatograph	Agilent Technologies Inc., Santa Clara CA, USA
flux integration		STAR	Varian

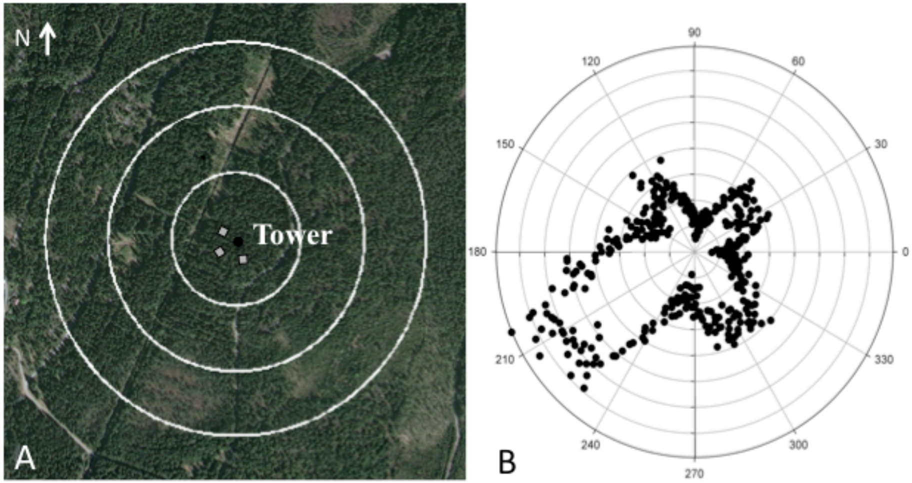

Fig. 1. (A) Position of the eddy covariance tower and of the three chamber measurement stations at the Skogaryd research catchment. White circles indicate the distance to the tower (100 m, 200 m, and 300 m, respectively) **(B)** Relative wind frequency for each wind direction (0–360°).

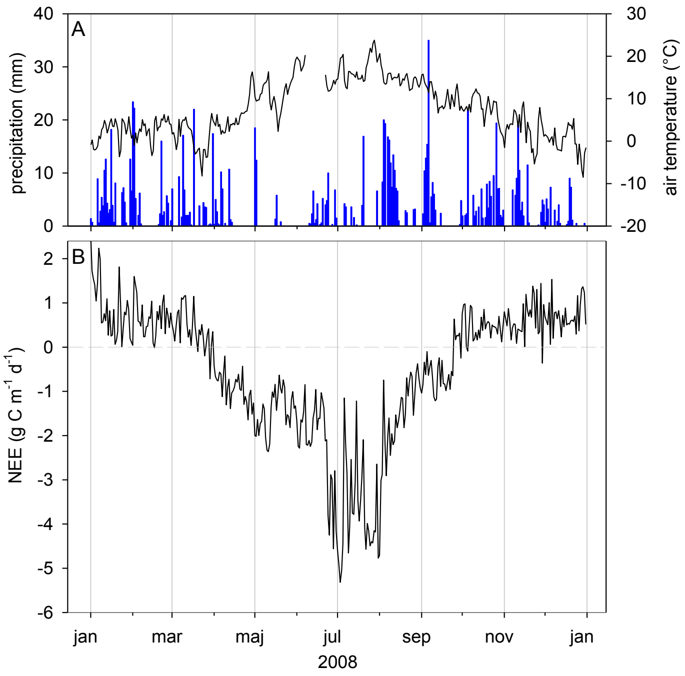

Fig. 2. Daily precipitation (mm) from the Vänersborg SMHI weather station and average daily air temperature (°C) **(A)** and net ecosystem exchange (NEE: g C m^{-2} d^{-1}) **(B)** at the Skogaryd research catchment.

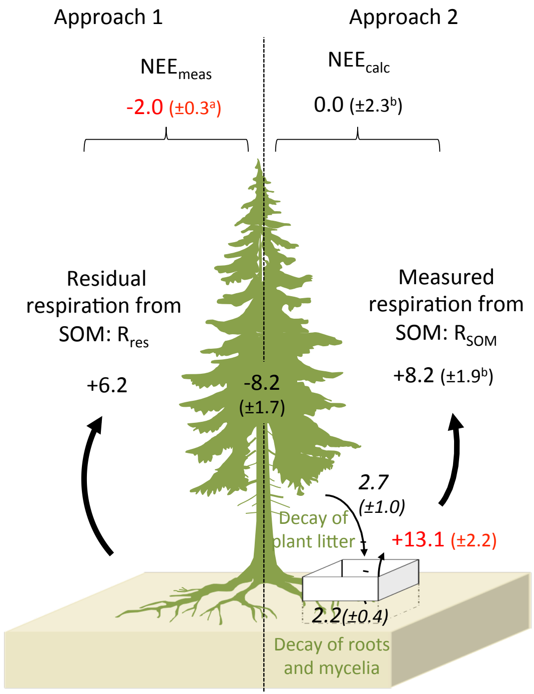

Fig. 3. Net ecosystem C exchange at the Skogaryd research catchment as determined by two approaches. In approach 1 (left), NEE is measured directly by the EC technique and the respiration from SOM (R_{res}) is determined as the residual flux of the measured NEE$_{\text{meas}}$ and the calculated tree growth. In approach 2 (right), the NEE is calculated as the residual flux NEE$_{\text{calc}}$ based on the quantification of the respiration from SOM (R_{SOM}, see text for details). Measured gross fluxes are given as red numbers, calculated gross fluxes as black numbers, calculated net effluxes as italics. All fluxes in $\text{t C ha}^{-1}\,\text{yr}^{-1}$. ^{a} Assuming 15 % data uncertainty, ^{b} propagated error estimates are calculated by the Gaussian error propagation scheme.