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

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Abstract

Afforestation has been proposed as a strategy to mitigate the often high greenhouse gas (GHG) emissions from agricultural soils with 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 8.3 ± 3.9 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 probably in its phase of maximum C assimilation or shortly after. The N_2O fluxes were determined by the closed chamber technique and amounted to 0.9 ± 0.8 t C_{eq} ha^{-1} yr^{-1}. According to the direct measurements from the eddy covariance technique, the site acts as a minor GHG sink of -1.1 ± 0.8 t C_{eq} 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 0.7 ± 4.6 t C_{eq} 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 accumulated uncertainties involved in the indirect approach, the direct approach is considered the more reliable tool. As the rate of C sequestration will likely decrease with forest age, the site will probably become a GHG source once again as the trees do not compensate for the soil C and N losses. Also forests in younger age stages have been shown to have lower C assimilation rates, thus the overall GHG sink potential of this afforested nutrient-rich organic soil is probably limited to the short period of maximum C assimilation.

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 (Couwenberg et al., 2011).

For Sweden, it has been estimated that organic soils used for agriculture release 6 – 10% of the total national emission of green-house gases (GHG) (Kasimir Klemedtsson et al., 1997; Berglund and Berglund, 2010), despite that only 5 - 10% of agricultural land is on such soils. These emissions are primarily a result of drainage, which increases soil aeration and consequently organic matter mineralization resulting in emissions of carbon (C) and nitrogen (N) rich 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 t C ha^{-1} yr^{-1} (Nykänen et al., 1998; Maljanen et al., 2004, 2007; see also review by Alm et al., 2007), but N_2O emissions can also be high (2–37 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., 2005b; 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 application 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).

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; Minkkinen et al., 2002; Hargreaves et al., 2003). 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 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., 2003b; von Arnold et al., 2005b). Thus, high soil C and N losses potentially outweigh C accumulation by the trees (Alm et al., 2007).

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). The Swedish National Inventory Report (NIR) (2012) submitted to the UNFCCC uses emission factors (EFs) for CO_2 of 3.0 and 1.9 t C ha^{-1} yr^{-1} for well drained and poorly drained forest soils, respectively, based on the work of von Arnold (2004). However, these emissions (in total 9.5 Tg) are subtracted from the uptake of CO_2 by the net forest production (29 TG), a much larger flux. As a consequence the soil emissions are not clearly made visible in the NIR. So far, N_2O emissions from drained organic forest soils have not been included in the Swedish NIR. This was decided by the Swedish Environmental Protection Agency according to the Conference of the Parties (No 9), arguing that reporting of N_2O emissions due to forest drainage is optional (UNFCCC decision 13/CP.9). However, the IPCC Good Practice Guidance (2003, www.ipcc-nggip.iges.or.jp/public/gp/english/) recommends to use EF's of 0.1 and 0.6 kg N_2O-N ha^{-1} yr^{-1} for nutrient poor and nutrient rich soils, respectively. The same default EF's from forests on drained organic soils are also included in the 2006 IPCC Guidelines. These will be the mandatory guidelines for future NIR accounting, and the default EF's are in the process of being updated by the IPCC Wetlands Supplement, which will be published in autumn 2013. However, great uncertainty exists with regard to the precision of the recommended N_2O EF's so far which is mainly due to the scarcity of available measurements, missing consideration of emissions predictors, and inaccuracies in up-scaling of emissions (von Arnold et al. 2005a). As a consequence, the calculation of total GHG budgets for drained organic forest soils based on IPCC Good Practice Guidance (2003) is rather unclear. 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). More data and a better process understanding are needed to fill this gap.

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 to determine the CO_2 flux and chambers measurements to quantify the N_2O and CH_4 fluxes, 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), which is why 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 for the soil C balance (Minkkinen et al., 1999) and for N_2O emissions of organic soils in the hemiboreal to boreal regions (Klemedtsson et al., 2005; Mäkiranta et al., 2007). ~~).~~ Klemedtsson et al. (2005) found a clear correlation between soil C:N and N_2O emissions, which 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, but this figure has not yet been used in the UNFCCC reports. However, the release of N due to mineralization is primarily governed by the water table level so that the drainage status is a key 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, has been shown to give different results depending on the equipment and chamber design used (Pumpanen et al., 2004; Christiansen 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. However, the NEE can only be measured 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. Most important, the technique does not work well under highly stable atmospheric conditions (Moffat et al., 2007; Vickers et al., 2012). The biggest problem with eddy covariance measurements is that they are conducted from usually only one tower, thus there is no replication. The source area (foot print) of the flux varies dynamically in relation to turbulent conditions; consequently foot print areas are generally smaller during days than during nights.

In view of the need to further quantify 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 Norway spruce since about 60 years. The GHG budget was determined for one year by estimating the ecosystem fluxes of N_2O, and CH_4 and the NEE. To quantify the NEE, two different approaches were used: 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 the basal area, BA) but Scots pine, *Pinus sylvestris* L. (13% of BA) and Silver birch, *Betula pendula* Roth (5% of BA), were also present. The dominant and co-dominant Norway spruce trees were 22 – 25 m in height and the average LAI was 5.1 m^2 m^{-2} at the bottom of the living crown in May 2007

(Dewar et al. 2012). 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 % between 15 – 40 cm. The site is well drained so that the average water table depth 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-year (1961–1990) annual mean temperature at the Vänersborg weather station, situated approximately 10 km east of the site, was 6.2 °C. The 30-year 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 (www.smhi.se). The daily 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.

A flux tower and three stations for soil flux measurements were established at the center of a relatively homogeneous area of the forest, extending about 200 m in all directions, except towards the E and SE (Figure 1a). The prevailing wind is from the SW (Figure 1b) where the drained afforested organic site extends for more than 300 m. The three soil flux stations are distributed within the footprint area of the EC tower. Each station consists of two plots. At each plot two measurement collars for each treatment (two control collars, two trenching collars, for details see below) were permanently installed on which the chambers for gas flux measurements were placed.

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 sensors for determination of soil temperature and water table depth were installed at two plots per station and were situated between the three measurement collars. Soil temperature was measured at soil depths of 0.05, 0.15, 0.30 and 0.60 m (n == 6 for each soil depth, in total fours depths). Water table depth was measured using submersible pressure transducers installed in plastic tubes inserted vertically 1.5m into the soil (n = 4). Soil temperature and water table depth were recorded every 30 minutes~~.~~. Details of the instruments used are given in the appendix (Table A1).

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 year global warming potential (GWP) of each gas (as given in the IPCC,~~;~~ Forster et al. 2007). The net flux of N_2O and CH_4 was determined from soil flux 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:

$$\mathrm{NEE} = \mathrm{GPP} - R = \mathrm{GPP} - R_a - R_h \tag{1}$$

$$\mathrm{NPP} = \mathrm{GPP} - R_a \tag{2}$$

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 equations (1) and (2) gives:

$$\mathrm{NEE} = \mathrm{NPP} - R_h \tag{3}$$

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:

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

where AGB_{inc} and 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.

The heterotrophic soil respiration R_h is typically determined by the soil chamber technique which measures the soil CO_2 efflux over time. In this study a trenching experiment was applied in order to separate between the CO_2 flux from roots/mycelia and SOM. As a consequence, the determination of the components of R_h, the respiration from decomposition of soil organic matter (R_{SOM}) and decomposition of leaf and root litter (R_{LL}, R_{LR}) become important:

$$R_h = R_{SOM} + R_{LL} + R_{LR} \qquad (5)$$

Combining equations (3), (4b) and (5) gives:

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

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, equation (6) can be simplified to:

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

So that

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

Equation (6a) is used in approach 2 to calculate the NEE (in the following $\mathrm{NEE}_{\mathrm{calc}}$), based on determination of AGB_{inc}, BGB_{inc} and R_{SOM}, while in approach 1, based on eddy flux measurements of the NEE (in the following $\mathrm{NEE}_{\mathrm{meas}}$), R_{SOM} was calculated according to equ ation (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 (average tree height: 20 m) were carried out using the EC technique (Grelle and Lindroth, 1996). The instrumentation used was basically the same as the system described in Lindroth et al. (2008) and is listed in the appendix (Table A1). 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 appendix). 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 analysis of the CO_2 concentration 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 tubing. 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 (In Situ Instrument AB, Ockelbo, Sweden). In particular, flux corrections for air density fluctuations ("WPL correction", Webb et al., 1980) were accounted for by converting the measured absorbance to dry mixing ratios by the Ecoflux software. Losses of high frequency information due to signal attenuation in the sampling tubes was corrected by transfer functions based on tube length and diameter, air flow rate, and diffusivity of CO_2 and H_2O, respectively. These transfer functions were then used to modify theoretical turbulence spectra based on wind speed, stability, and measurement height (Kaimal et al., 1972). From the ratio between original and modified spectra a correction factor for signal attenuation was then estimated for each 30-minutes interval.

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 technically flawless and CO_2 flux spikes were removed which could not be attributed to morning flushes. The CO_2 storage term was calculated using data from the measurements of the vertical CO_2 gradient added to the flux data to obtain the NEE. The annual average CO_2 storage flux was on average 2 % of the EC flux.

The gap filling was conducted as follows: Daytime and night time data were selected according to the u* > threshold, i.e. the friction velocity at which the eddy covariance turbulence criteria, i.e. the friction velocity at which the mixing was well developed, are fulfilled. The night time 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 s^{-1}.

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

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

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

Night time data from each period were fitted to the function:

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

where F_{cn} is the night time ~~biotic~~ NEE flux (µmol 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 night time 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.

Gap filling error estimates were done by randomly removing a certain number of measured values, filling them according to the gap filling method used, and then comparing the estimated gap-filled values with the measured values. This procedure was repeated 10 times, and the mean gap filling error was estimated for day and night values.

The flux footprints (or flux source areas) were estimated based on the algorithms of Schmid (1994, 2002). For each 30-minutes average interval, the dimensions of the elliptical source area from which 80% of the flux emanated were calculated. The corresponding surface areas were then aggregated for one year in a polar coordinate system centered by the position of the tower, resulting in three dimensional maps displaying the frequency of flux origins for each surface element within the fetch of the tower, separately for summer and winter, day and night, respectively. For simplicity, footprints from all wind directions were then accumulated and normalized into single graphs showing the frequency of flux emanation as a function of distance from the tower.

Because of the effect of atmospheric stability, flux footprints are generally larger and further away from the tower during night-time and during winter than during day and summer.

However, even during winter nights more than 90% of the fluxes emanated from within a 175-m radius around the tower.

2.5 Quantification of above and belowground tree growth

The annual production of aboveground and belowground biomass (including roots > 1cm) (eq.6a, 6b: AGB_{inc}, BGB_{inc}) was determined using biomass functions developed by Marklund (1988) and modified by Minkkinen et al. (2001) for organic soils, based on diameter at breast height (*dbh*) and tree height. These parameters were measured for 134 trees from 16 circular sub-plots of 0.01 ha (in total 0.16 ha) distributed within a rectangle around the EC tower. The *dbh* was measured at the turn of September and October in 2007 and 2008. The initial height of the trees in in 2007 was estimated based on *dbh* 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 2010 on 335 trees in a 90 × 90 m plot surrounding the EC tower (Figure 1a). The 16 small sub-plots of 0.01 ha sampled for *dbh* were distributed uniformly within this area. The tree height increment in the 16 sub-plots was calculated from the mean annual increment in height, calculated as:

$$h_{inc} = \frac{h_{tot} - 1.3}{a_{tot} - a_{dbh}} \qquad (9)$$

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

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

Since the 16 sub-plots were found to have on average a 23% smaller BA than the larger plot surrounding the EC tower based on an inventory of all 797 trees of the larger 90 x 90 m plot (Figure 1) in 2010, the biomass production was adjusted accordingly. A function established from a regression analysis of annual growth increment versus BA ($R^2 = 0.5$) for the 16 sub-plots was used to recalculate the biomass production by applying the BA of the larger plot on this function and assuming an annual *BA* increase of 1.1 m^2 ha^{-1} (Klemedtsson et al., 2010) to adjust for changes between 2008 and the 2010 inventory.

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 biomass, that was was determined from the Minkkinen et al. (2001) equations, which amounted to 3.6% of the total biomass increment. This assumption was based on publications by He et al. (2012) and White et al. (2000) showing that the ratio between the annual C increment of fine roots and of needles ranges between 0.66 and 3.3 kg C kg C^{-1} for conifers. The majority of given values for spruce-fir forests lay between 1 and 1.5 kg C kg C^{-1}. Due to the wide range of values, we assigned an error estimate of 300% to the fine root growth.

In order to receive an uncertainty estimate of tree growth, we quantified the measurement uncertainty of *dbh* and tree height. For this, we calculated the respective biomass in 2007 and 2008 by assuming an error of ± 5% in *dbh* and of ± 10% in tree height. The respective values for error estimates were taken from published data (McRoberts et al. 1994). The measurement uncertainty in tree growth was then determined by the Gaussian error propagation scheme and the respective errors of the biomass in 2007 and 2008 (eq. 14). Furthermore, an uncertainty was assigned to the correction factor used for considering the 23% smaller BA of the 16 small plots compared to the large 90 x 90m plot. The error estimate was determined 50% since the variation in BA explained 50% of the variation in tree growth ($R^2 = 0.5$)

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_{CO2}) 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_{CO2} originates from three different sources, the flux from SOM, the flux from trenched roots and mycelia, and the flux from the litter layer (litter from aboveground plant parts)~~:~~

$$E_{CO2} = R_{SOM} + R_{decay} + R_{LL} \quad (10)$$

So that

$$R_{SOM} = E_{CO2} - R_{decay} - R_{LL} \quad (10a)$$

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_{CO2} was measured using automatic transparent closed chambers at the three stations comprising two measurement plots each (n = 6) within the footprint area of the EC tower (Figure 1a). Details about the chamber design and instrumentation can be found in the appendix. Chambers were placed on permanently installed collars that covered the trenching frames. All vegetation was removed to eliminate CO_2 uptake by photosynthesis. A logger system connected to the chambers prompted a measurement cycle every 20 minutes. 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. The CO_2 concentration of the chamber air was averaged every 30 seconds for 15 minutes before the lid was opened again and the cycle restarted. Measurements run automatically all day for a period of three weeks at one station. Chambers were then moved to the next station where another measurements cycle of three weeks was started.

The respiration 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:

$$R_{decay} = (1-\alpha) \cdot m \cdot (1 - e^{-0.997 \cdot k_r}) \quad (11)$$

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 respiration from the litter layer, R_{LL} that contributes to the measured soil efflux E_{CO2} (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:

$$R_{LL} = L_l - \Delta L \qquad (12)$$

Where L_l = average annual aboveground litter production (t C ha^{-1} yr^{-1}), and ΔL = average annual accumulation of C in the litter layer (t C ha^{-1} yr^{-1}).

In order to estimate ΔL, a total of 30 litter layer 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 m^2; n = 10) that were placed evenly across the study area. Traps were emptied every sixth week, all litter fractions (needles, twigs, cones, branches) were considered and the C content was determined from sub-samples.

2.7 Soil N_2O and CH_4 fluxes

Besides the CO_2 fluxes, 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. Measurements were taken at the in total six control plots of the three stations around the EC tower (n=6) by means of a dark chamber with a volume of 0.017 m^3 that was placed on the measurement collars (0.19 m^2) (see also section 2.1). Gas samples from the chambers were taken 4, 8, 16 and 32 min. after lid closure and collected in vials. Gas concentration was analyzed by gas chromatography and the gas flux was determined from the slope of the linear 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. Further details on sample taking, gas analysis and flux calculation can be found in Ernfors et al. (2011).

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.

For E_{CO2} a simple linear regression between hourly measured E_{CO2} and soil temperature at 30 cm depth was fitted for each plot according to the function:

$$E_{CO2} = a + b \cdot t_{30} \qquad (13)$$

where a and b are fitting parameters and t_{30} is the soil temperature at 30 cm depth.

Including water table level in the regression was tested but did either not improve goodness of fit or led to a negligible increase. Agreement between model parameters was very ~~good~~high for the plots of stations 1 and 2 so that the same model could be used whereas for station 3 other parameters values had to be applied. Variation in soil temperature explained on average more than 75% of variation in E_{CO2}. Annual fluxes were calculated by summing up modelled daily values. Standard deviation of E_{CO2} is given as the variation between stations 1, 2 and 3.

Also for N_2O fluxes a simple linear regression model to soil temperature and water table depth was tested using daily mean values. Regression –was very low except for station 3 (pearson r below 0.4)~~,~~. Thus, no regression model was applied to extrapolate fluxes. To calculate annual flux balances, mean daily N_2O and CH_4 fluxes of the missing periods were interpolated between two consecutive measurements. Annual flux balances were derived by summing up daily fluxes for each station which were then converted to C equivalents using the GWP for a 100 year time frame of 298 (N_2O) and 25 (CH_4) and the respective molar masses of CO_2 and C (IPCC: Forster et al., 2007). To quantify the uncertainty induced by gap-filling of N_2O fluxes, we calculated the gap-filling uncertainty for each gap-filled day by taking the~~-~~ respective standard deviation of the values used to fill the gap into account. The total flux uncertainty was quantified according to the Gauss error propagation scheme by using the standard deviation and the gap-filling uncertainty of each station (eq. 14).

Generally, the Gauss error propagation scheme was used to calculate total errors of variables consisting of several compounds (tree growth, NEE_{calc}, R_{SOM}, R_{res}, N_2O *flux*-) assuming that the respective components (as given e.g. in eq. 6a and 10a) -were independent:

$$SE_{tot} = \sqrt{SE_{c1}^2 + SE_{c2}^2 + SE_{cx}^2} \qquad (14)$$

where SE_{tot} = propagated error, and SE_{cx}= standard deviation of component 1…x.

3 Results

3.1 Abiotic data

The year 2008 was considerably warmer (8.7 ˚C) than the 30 year 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 (Figure 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 nearby weather station (Vänersborg) was above average (10-year 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-year average for that period (177 mm). A persistent snow layer of up to 35 cm was observed for only two weeks in March (www.smhi.se).

Annual average soil temperature varied across the site and was 4.9 °C, 5.8 °C and 6.9 °C for station 1, station 2, and station 3, respectively. Annual average water table depth for the whole site was 0.45 m- with only small variations between stations: 0.15 m to 0.85 m (Figure 2b) ~~.~~

3.2 NEE as determined by micrometeorological methods

The accumulated NEE in 2008 was -2.0 ± 0.3 t C ha^{-1} yr^{-1} and showed large seasonal variation. During the first winter period NEE ranged between 0 and 1.5 g C m^{-2} d^{-1} with a maximum release of 2.3 g C m^{-2} d^{-1}. The NEE was positive from January 1st until March 29th when the ecosystem switched to being a C sink (Figure 2b). NEE decreased rapidly until mid-May but remained then constant until the end of June. The highest net CO_2 uptake was reported in July (-5.3 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 g C m^{-2} d^{-1}. In November and December the NEE showed a strong negative correlation to air temperature (r^2 = -0.78).

The average gap filling error was estimated to 6 % and 14 % for daytime and night time values respectively, corresponding to an uncertainty of ± 13 g C m^{-2} for all gap-filled data. This corresponds to an uncertainty of ± 6.5% of the total annual flux. The gap filling procedure showed good agreement during the growing season although the night time fluxes were underestimated. During winter the night time fluxes were heavily overestimated by the gap filling model, although it did not contribute considerably to the total annual flux.

During daytime, the source distance distribution around the tower peaked at 50-100 m and the 90 % limit was within 200 m both during daytime and night time (Figure 1c). During night time, the source area occasionally moved further away, but never more than 250 m. 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 (Figure 1b).

3.3 Biomass production

Standing biomass (including roots ≥ 1 cm) – calculated using the Minkkinen et al. (2001) equations – amounted to 259 and 273 t dm ha^{-1} in 2007 and 2008, respectively. The coarse root biomass (> 1 cm) was estimated to be 45.7 t dm ha^{-1} (2007) and 47.9 t dm ha^{-1} (2008). The resulting tree growth in 2008 corrected according to the BA factor and including fine root growth was 8.3 ± 3.9 t C ha^{-1} yr^{-1}.

3.4 NEE as determined by compartment fluxes

Applying eq. (6a) to the measured data of soil processes and biomass production resulted in a NEE of 0.2 ± 4.5 t C ha^{-1} yr^{-1} (Table 4). The measured soil CO_2 efflux, E_{CO2}, in 2008 was 13.0 ± 1.7 t C ha^{-1} yr^{-1}. Fluxes were highest from station 3 (14.9 t C ha^{-1} yr^{-1}) whereas fluxes from the other stations were 12.6 (station 1) and 11.5 (station 2) t C ha^{-1} yr^{-1}. The reason behind the variation is likely the higher average soil temperature at station 3 since it could be shown by the regression analysis that measured heterotrophic soil efflux E_{CO2} responded strongly to soil temperature whereas the inclusion of water table depth only slightly increased the fit of the regression model. The low response to water table depth was also observed by Ojanen et al. (2010) and Mäkiranta et al. (2008) and is likely to be caused by the generally high drainage level of below 30 cm. Fresh litter with the highest potential for decomposition is deposited on the soil surface so that the water table would need to rise considerably to have an effect on its decomposition.

The respiration from decaying roots in the trenched plots amounted to 1.5 ± 0.3 t C ha^{-1} yr^{-1} for fine roots and 0.7 ± 0.5 t C ha^{-1} yr^{-1} for coarse roots, resulting in a total R_{decay} of 2.2 ± 0.4 t C ha^{-1} yr^{-1} (Table 1). The biomass of mycorrhizal fungal mycelia was very small, which is why their impact on the CO_2 flux was ignored. The average annual litter fall was 2.9 ± 1.1 t C ha^{-1} yr^{-1} and the annual litter accumulation 0.2 ± 0.1 t C ha^{-1} yr^{-1}, which sums to an annual net respiration from decomposing litter of 2.7 ± 1.0 t C ha^{-1} yr^{-1}. As a result, the respiration from SOM, R_{SOM}, was 8.1 ± 2.1 t C ha^{-1} yr^{-1}.

3.5 N_2O and CH_4 fluxes

The measured CH_4 flux was, on average, -4.4 ± 2.3 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 ± 9.3 kg N_2O ha^{-1} yr^{-1} (see also Ernfors et al., 2011) which corresponds to 0.9 ± 0.8 t C_{eq} ha^{-1} yr^{-1} (Table 2). The response of N_2O fluxes to soil temperature and water table depth was low for all plots expect one ($R^2 < 0.4$), so that no regression model was applied to extrapolate fluxes.

3.6 Two approaches to determining NEE and overall GHG budget

The measurements conducted in this study allow for 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 Figure 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 therefore likely to exhibit 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 soil efflux, corrected for the CO_2 efflux from decaying roots and litter. The respiration from SOM (R_{SOM}) was 8.1 ± 2.1 t C ha^{-1} yr^{-1} so that the NEE_{calc} became -0.2 ± 4.5 t C ha^{-1} yr^{-1} (Figure 3).

When NEE was combined with fluxes of CH_4 and N_2O (see above) the total flux balance of GHGs using approach 1 and 2 became -1.1 ± 0.8 t C_{eq} ha^{-1} yr^{-1} and 0.7 ± 4.6 t C_{eq} ha^{-1} yr^{-1}, respectively (Table 4). Thus, the GHG budget was almost in balance, but approach 1 indicated a small net sink while approach 2 indicated a small net source.

4 Discussion

4.1 Quantification of the NEE and compartmental fluxes

The two approaches used to quantify the NEE gave different results. The uncertainty associated with NEE_{calc} is much higher than the uncertainty associated with NEE_{meas}, although it must be kept in mind that the propagated error of 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 of other similar sites is hampered because of the scarcity of year-round measurements in the literature. To our knowledge, there are no similar studies from Sweden, but in Finland studies at similar site conditions (depth of the peat, vegetation, drainage level) and climate conditions (situated in the hemiboreal to boreal region) have been conducted. Lohila et al. (2011) found a NEE of -2.4 t C ha^{-1} yr^{-1} for a pine forest on a drained organic soil (Table 3). However, that site had no agricultural history and the 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 site of the present study, a net accumulation of C in the soil due to limited peat decomposition. A young pine forest on former agricultural land was found to be a restricted C source of 0.5 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 years old with a C accumulation of 2.4 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 site possibly could be a C sink if C accumulation in ground vegetation and belowground litter production were included. Other studies on the NEE of forests on organic soils have been undertaken. by Dunn et al. (2007) and Flanagan and Syed (2011), but the comparability is limited, because site conditions differ considerably from our site with respect to vegetation (natural, not afforested), management history (no drainage) and the much higher forest age of 160 years. Overall, the NEE_{meas} from our study appears to be reasonable compared to values reported from Finland, whereas the NEE_{calc} seems comparatively high considering the high level of C accumulation in trees.

The estimated tree growth of 8.3 ± 3.9 t C ha^{-1} yr^{-1} is higher than reported values for similar ecosystems in Sweden and 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. (2005b) can probably be explained by tree age. The trees at their site were 90 years old, and were already older than the age associated with maximum biomass production (Gower et al., 1996; Black and Gallagher, 2010).

The high C uptake from the trees is to a large extent (approach 1) or completely (approach 2) compensated by the respiration from SOM. However, when determined by the chamber technique (R_{SOM}, approach 2) respiration was about 30% higher compared to the residual flux R_{res} of approach 1. R_{SOM} is higher than reported values from organic sites across Finland and Sweden (-0.8 to 6.7 t C ha^{-1} yr^{-1}, see Table 3) and also higher than fluxes from nutrient rich agricultural sites (4.5 to 5.5 t C ha^{-1} yr^{-1}, Kasimir Klemedtsson et al., 1997) whereas R_{res} ranges at the upper end of reported values. Generally, the heterotrophic soil CO_2 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). The discrepancy between approaches points to some uncertainties associated with the methods applied which will be discussed in detail in section 4.5.

4.2 Potential 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 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 kg C ha^{-1} yr^{-1}, Öquist et al., 2009, Rantakari et al., 2010; Wallin et al., 2010; Dyson et al., 2011). Overall, with reported maxima of 30 and 115 kg C ha^{-1} yr^{-1}, the contribution of discharge C losses to the total C budget at Skogaryd is probably low as literature values constitute less than 2% of the annual tree growth and the respiration from SOM

4.3. N_2O and CH_4 fluxes

The measured CH_4 flux was very low and always negative, i.e. CH_4 was taken up by the soil. These results are similar 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 kg C ha^{-1} yr^{-1}) (e.g. Kasimir Klemedtsson et al., 1997; Minkkinen et al., 2002; Huttunen et al., 2003; Maljanen et al., 2003a; von Arnold et al., 2005b; Yamulki et al., 2013~~2~~). Due to the low magnitude of the flux, the contribution to the GHG budget is negligible (Table 4). 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., 2005b; Minkkinen and Laine, 2006).

In contrast to CH_4, significant N_2O fluxes of 11.1 ± 9.3 kg N_2O ha^{-1} yr^{-1} were observed which are higher than reported values (0.4–8.1 kg N_2O ha^{-1} yr^{-1}: Maljanen et al., 2003b; von Arnold et al., 2005b; Alm et al., 2007; Yamulik et al., 2012). Comparing the 2008 flux estimate to the preceding and following year (2007: 9.7 ± 8.9 kg N_2O ha^{-1} yr^{-1}, 2009: 5.1 ± 2.6 kg N_2O

ha^{-1} yr^{-1}) shows that fluxes were above the three-year average of 8.5 kg N_2O ha^{-1} yr^{-1} although similar to the values in 2007. In 2008, also air temperature and precipitation were above average, so that especially in August and September warm and wet conditions have stimulated N_2O emissions (data not shown). Therefore, fluxes in 2008 were exceptional, but reasonable with regard to climatic conditions. The in general high N_2O emissions from the present site can be explained by the relatively low 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 ratio of 25. Although water table has been identified as one of the determining factors for N_2O emissions, especially for oligothrophic soils, (Martikainen et al., 1993) the response of N_2O fluxes to the water table depth in our study was only low as tested by a regression model. The same applies to the soil temperature. One reason behind is probably the only fortnightly measurement frequency. Furthermore N_2O emissions seem to respond strongly to intermediate soil moisture conditions (Jungkunst and Fiedler, 2007), which means that changing water table depths creating frequent changes between drier and wetter conditions could stimulate N_2O production so that no clear correlation can be found.

Our numbers corroborate the N_2O effluxes summarized in a review by Alm et al. (2007), where an efflux of up to 10 kg N_2O ha^{-1} yr^{-1} was reported for nutrient-rich well-drained pine fens and spruce mires. Together, these results indicate that the value of the default emission factor in the 2006 IPCC Guidelines for nutrient rich drained organic forest soils are probably too low: 0.9 (0.25–3.7) kg N_2O ha^{-1} yr^{-1}.

4.4. Overall GHG balance

Considering all GHG fluxes, the calculated GHG budget is negative (-1.1 ± 0.9 t C ha^{-1} a^{-1}) when determined according to approach 1 or positive when determined by approach 2 (0.8 ± 2.8 t C ha^{-1} a^{-1}, Table 4). The GHG budget estimations by the two approaches differ by 2 t C ha^{-1}yr^{-1} but the high uncertainties associated with them do not allow for concluding that they, in fact, are different. Our results contrast with the findings of Ojanen et al. (2012) who compared the EC technique and the chamber technique and balance calculations to determine soil C balance. They found only marginal differences between the soil effluxes using the two different methods. 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 EC and the chamber approach by Ojanen et al. (2012) differed by almost 2 t C ha^{-1} yr^{-1}. Furthermore they did not, to our knowledge, correct for the decay of roots and mycelia ~~v~~in trenched measurement collars. 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.

The difference between approaches and the high uncertainty of NEE_{calc} point to some potential uncertainties which are associated with the method used to estimate the respective compartmental fluxes.

4.5 Uncertainty in the terms of the C balance

4.5.1 Biomass

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 can in principle be distinguished between model uncertainty, sampling uncertainty and measurement uncertainty (Melson et al. 2011; Chave et al. 2004; Jenkins et al., 2003). Model uncertainty is caused by an inappropriate set of allometric functions and parameters to describe the forest stand and its' species composition accurately. However, ~~I~~in our study, the application of the Minkkinen et al. (2001) equations is the most appropriate model available. First, the equations were specifically developed for species common in Swedish forests, i.e. for the species studied (Norway spruce, Scots pine and silver birch), 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). Sampling uncertainty is related to the size of the study plot. In this study the systematic selection of 16 plots (0.01 ha each) in a grid around the EC tower, was shown to underestimate the BA compared to a measurement of all trees in a larger area (90 x 90 m) around the EC tower and could be seen as a random error. We corrected this effect by applying a regression model and introducing a correction factor which was assigned an uncertainty corresponding to the coefficient of determination. Furthermore, the uncertainty induced by measurement errors was quantified by assuming an error in *dbh* and tree height of 5% and 10%, respectively. The overall uncertainty of the annual tree growth amounted to almost 50% of the tree growth estimate: 3.9 t C ha^{-1} a^{-1} of which the largest part is attributed to the measurement uncertainty.

4.5.2 Approach 1: quantification 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, most errors are generally small. Based upon our own experience, errors related to gas analyzer calibration and signal dampening in tubes are both typically within 2-3%, while errors due to sensor separation are less than 2% (Berger et al., 2001). Random errors in EC measurements are generally considered to be less than 7% during daytime and less than 12% during night time 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 night time 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). We estimated the uncertainty introduced by the gap filling to be ± 13 g C m^{-2} on an annual basis which is in the same order of magnitude as the error estimated by Moffat et al. (2007) (±25 g C m^{-2} $year^{-1}$).

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, since the source area was determined to be by 90% within 200 m in all directions of the tower and potentially moved further away only occasionally at night time (at maximum 250 m), the potential influence of fetches of different vegetation and soil properties could only have been minor. Thus, considering the errors in calibration, spectral correction, gap filling and the fact that the footprint was well within the homogeneous part of the forest, we estimated the total error of the NEE measurements to be within ± 15% which gives a range of 1.7 to 2.3 t C ha^{-1} $year^{-1}$.

Whereas in approach 2 the respiration from SOM was determined according to chamber measurements (and balance calculations), in approach 1 respiration from SOM was determined as the residual between NEE_{meas} and tree growth. For both approaches, the fluxes differed by almost 30%. The uncertainty estimates of both fluxes demonstrate that they agree to each other within their uncertainty range. However, besides the errors induced by the respective flux components, the discrepancy in the average flux might be induced by a mismatch of the measurement period between flux measurements and tree growth estimation. Tree growth in spring is often initiated by re-mobilization of stored carbohydrates which were assimilated during the preceding autumn. This could imply that the measurement periods of NEE and chamber fluxes and tree growth do not overlap precisely which would lead to a bias in the estimation of R_{res}. Barford et al. (2001) and Gough et al. (2008) assign this time lag as reason for the bad agreement between biometric and flux measurement for short time periods but for the good agreement when long periods are considered. Thus, when short term periods are considered as in this study, this potential mismatch cannot be excluded.

4.5.3 Approach 2: On the indirect quantification of the NEE

The magnitude of NEE_{calc} in approach 2 is heavily based on precise quantification of the respiration from SOM ($R_{SOM.}$). Due to the use of a trenching experiment, the measured soil CO_2 efflux (E_{CO2}) had to be corrected for the CO_2 flux from the litter layer (R_{LL}) as well as the decay of roots (R_{decay}, ~~(~~Table 1). This correction accounts for 38% of the measured efflux.

Uncertainties in the CO_2 flux from the litter layer are difficult to evaluate. The assumption of a linear accumulation of litter since afforestation potentially induces a bias since year to year accumulation is likely to differ and accumulation rather follows a linear curve after canopy closure. Calculating litter accumulation since canopy closure (assumed to occur 10, 15, 20, and 25 years after afforestation) changed the average annual accumulation between 0.34 and 0.9 t C $ha^{-1}a^{-1}$. Another source of uncertainty is induced by a potential measurement error in the litter layer C storage. However, assuming an error of ± 20% in the measured dry weight only had a minor effect on the calculated total respiration from the litter layer: R_{LL} ranged between 2.7 ± 1.1 and 2.77 ± 1.1 t C ha^{-1} yr^{-1}.

An alternative method applied previously to quantify the respiration from SOM is to remove the litter layer before the experiment and install a mesh to exclude fresh litter (Mäkiranta et al., 2007). However, besides generating uncertainty in quantifying the amount of C removed this methods provokes 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). The calculated fine root mass loss was 25% of the initial mass for the first year after trenching. Comparability to studies from the boreal zone is limited due to the scarcity of studies but is within the range reported in studies from other zones, e.g. Epron et al. (1999) for a beech forest in France (26%), Diaz-Pinez et al. (2010) for a mixed spruce forest in Austria (30% in the first two years) (see also review by Subke et al., 2006). An alternative approach to estimate the impact of trenched roots was applied by Comstedt et al. (2011). Using a modeling approach based on a trenching experiment, they quantified the overestimation of the measured respiration 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 water table depth, 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 change of the water table depth after trenching, so this effect will have been minor.

4.5.4 On N_2O and CH_4 fluxes

The uncertainty estimate of the average annual N_2O flux constituted about 85% of the flux itself. To this, the standard deviation between stations contributes by more than 50% which points to the challenge of measuring N_2O fluxes in a heterogeneous environment like organic soils. Only a high measurement frequency and high spatial resolution could narrow this uncertainty. Also, the fortnightly measurement frequency might provoke that certain emission peaks are not captured. However, this would mean that the actual N_2O flux estimate of 11.0 ± 9.3 kg N_2O ha^{-1} is too low. A further bias of the N_2O flux estimate introduced by the fortnightly measurement frequency is the interpolation of measured fluxes for missing periods. We assessed this uncertainty by assuming a standard deviation of the daily gap-filled flux corresponding to the standard deviation of the data used to fill the gap. This bias contributed by about 40% to the total error.

One concern with regard to the application of the chamber technique is that the technique itself creates artifacts (Livingston and Hutchinson, 1995) which are discussed in detail elsewhere (e.g. Pumpanen et al., 2004; Forbrich et al., 2010; Levy et al. 2011). A lack-of-fit of the regression model for the increase of gas concentration over time in the measurement chambers is considered to be the largest source of uncertainty and especially the use of a linear model has been proposed to underestimate fluxes (Levy et al., 2011). Also, the use of a chamber without ventilation seems to underestimate fluxes (Christiansen et al., 2011). In contrast to these findings, 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 agreed very well to each other. It must be emphasized, that a potential bias caused by the assumption of a linear regression and the use of a chamber without ventilation again rather point to a potential underestimation of fluxes.

4.6 Comparison of approaches and long-term GHG balance

Although the average NEE determined by the two approaches differs by 2 t C ha^{-1}, both approaches are consistent given the large uncertainty associated with NEE_{calc} of approach 2. Due to the limitations of the methodology applied for approach 2 as discussed above, we consider approach 1, the EC technique, to be the more reliable tool in this case to obtain precise estimates of the ecosystem C budget although comparability between the biometric tree growth measurements and the EC methodology to receive an estimate of R_{res} is limited due to the time lag between CO_2 assimilation by the trees and tree growth as discussed in Gough et al (2008) and Barford et al. (2001). The study demonstrates, that future research is needed to narrow uncertainties related to the approach 2, especially with regard to the determination of SOM respiration by the chamber technology. Although the approach is too uncertain to provide a precise quantification of the GHG budget, the analysis of the compartmental fluxes provides estimates of their magnitude and can improve process understanding~~.~~

Approach 1 indicates that the study site is a GHG sink of -1.1 ± 0.9 t C_{eq} $ha^{-1}a^{-1}$ and that the N_2O efflux offsets about 50% of the C sink strength (Table 4). Although the N_2O flux estimate is associated with high uncertainty induced by the heterogeneous soil conditions and gap-filling, the comparison to the previous and following year show, that fluxes were only slightly above average which could be explained by climatic conditions. Furthermore, additional biases, which are difficult to assess, would lead rather to an underestimation than an overestimation of the flux.

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, can give some indication:~~.~~ Using snap-shot measurements and modeling, Hargreaves et al. (2003) showed that young forests of about 4 to 8 years 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. Gower et al. (1996) estimated the age of boreal evergreen forests associated with maximum biomass production to about 60 – 70 years. 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 (Gower et al. 1996; 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. Furthermore, the observed and estimated N_2O and CO_2 fluxes from the soil range at the upper end of or are higher than reported values. This implies that the overall GHG balance is very sensitive to changes in the tree C accumulation and that minor changes in the C assimilation could turn the ecosystem into a GHG source.

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., 2005b; Alm et al., 2007) and can compensate or 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.

5 Conclusion

We investigated the total GHG budget of a spruce forest on a nutrient-rich organic soil using the chamber technique to quantify N_2O and CH_4 fluxes and applying two approaches to determine the C balance: an indirect balance calculation approach based on chamber measurements and the direct measurement of the NEE by eddy covariance. Due to accumulated uncertainties that remains to be resolved using 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 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.

APPENDIX A

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	Manufacturer
ground abiotic data: recorded every 30 minutes			
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
water table level		CS-420-L Submersible pressure transducer	Campbell Scientific, Logan UT, USA
logger		CR10X logger, AM16/32A multiplexer	Campbell Scientific, Logan UT, USA
EC flux system			
CO_2, H_2O concentration	27	infrared gas analyser LI-6262	Li-Cor Inc., Lincoln USA
3D wind speed, air temperature	27	sonic anemometer Gill R3	Solent, UK
meteorological data: recorded every 10 minutes			
air temperature, air humidity	22	MP101A probe in ventilated radiation shield	Rotronic, 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
logger		CR10X logger	Campbell Scientific, Logan UT, USA

Table A1 (continued)

Parameter/ technical equipment	Instrument height/depth (m)	Instrument	Manufacturer
Soil respiration: automatic closed chambers (0.24m^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 minutes after chamber closure by means of extracting a gas sample with a syringe inserted into a rubber septum in the chamber lid. Chamber volume: 0.017 m^3			
determination of gas concentration of samples		Varian 3400 gas chromatograph	Agilent Technologies Inc., Satna Clara CA, USA
flux calculation		STAR	Varian

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Table 1 Biomass in autumn 2007 (t C ha^{-1}) and respective parameter values of fine roots, coarse roots, and fungal mycelia used for the calculation of the respiration 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 (t C ha^{-1})	Determined by method	α	k_r (yr^{-1})	Calculated flux (± stand. dev.) (t C ha^{-1})
Fine roots (< 1cm)	6.2	measured[a]	0.4 [b]	0.5	1.5 (0.8)[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)

[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 C equivalents (t C_{eq} ha^{-1} yr^{-1}) (±~~stand.dev.)~~uncertainty) at the Skogaryd research catchment for 2008. C equivalents were calculated based for a 100 year time frame based on their respectively global warming potential (IPCC: Forster et al., 2007)

	station 1	station 2	station 3	mean flux	C equivalents (tC_{eq} ha^{-1} yr^{-1})
N_2O flux (kg N_2O ha^{-1} yr^{-1})	3.5	11.1	18.7	11.0 ± 9.3	0.9 ± 0.8
CH_4 flux (kg CH_4 ha^{-1} yr^{-1})	-5.8	-1.7	-5.5	-4.4 ± 2.3	-0.08 ± 0.04

Table 3 Annual tree growth, NEE and respiration from SOM determined at comparable organic sites. If not stated otherwise, units are t C ha^{-1} yr^{-1}. A negative sign indicates an uptake from, a positive a release to the atmosphere.

Reference	Site	Period	Species	Forest age (yr)	Stem density (trees ha^{-1})	Tree growth	NEE	SOM respiration
von Arnold et al. 2005b	Sweden, Asa	2000-02	Norway spruce	90	750	3.5		
Lohila et al. 20011	Finland, Kalevansuo	2005	natural pine and birch		835	1.6	-2.4	
Lohila et al. 2007	Finland, Alkkia	2003	Scots pine	ca. 30	977	2.4	0.14	4.8
Mäkiranta et al. 2007	Finland, Alkkia	2003-04	Scots pine	32				1.3
Mäkiranta et al. 2010	Finland, Padasjoki	2001-04	Scots pine	60		1.9		4.1 to 2.6^{a}
Ojanen et al. 2010	Finland, 68 sites	2007-08	Coniferous					1.5 to 6.7
This study	*Sweden, Skogaryd*	*2008*	*Spruce dominated*	*60*	*985*	*8.3*	*-2.0^{b} to -0.2^{c}*	*6.3^{b} to 8.1^{c}*

[a] *For the years 2001 to 2004*
[b] *Approach 1: EC*
[c] *Approach 2: chambers*

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

		Approach 1		**Approach 2**	
		(GHG flux ± stdev)		(GHG flux ± stdev)	
Plant CO_2 flux					
Tree growth	ΔB	-8.3	3.9	-8.3	3.9
Soil CO_2 fluxes					
Measured soil flux	E_{CO2}			13.0	1.7
Respiration from litter layer	R_l			2.7	1.1
Decay from roots	R_{decay}			2.2	0.4
Soil net efflux	R_{res}/R_{SOM}	6.3	4.0	8.1	2.1
Net ecosystem CO_2 flux	NEE	-2.0	0.3	-0.2	4.5
Other GHG					
CH_4 flux		0.0	0.0	0.0	0.0
N_2O flux		0.9	0.8	0.9	0.8
total GHG flux balance		-1.1	0.8	0.7	4.6

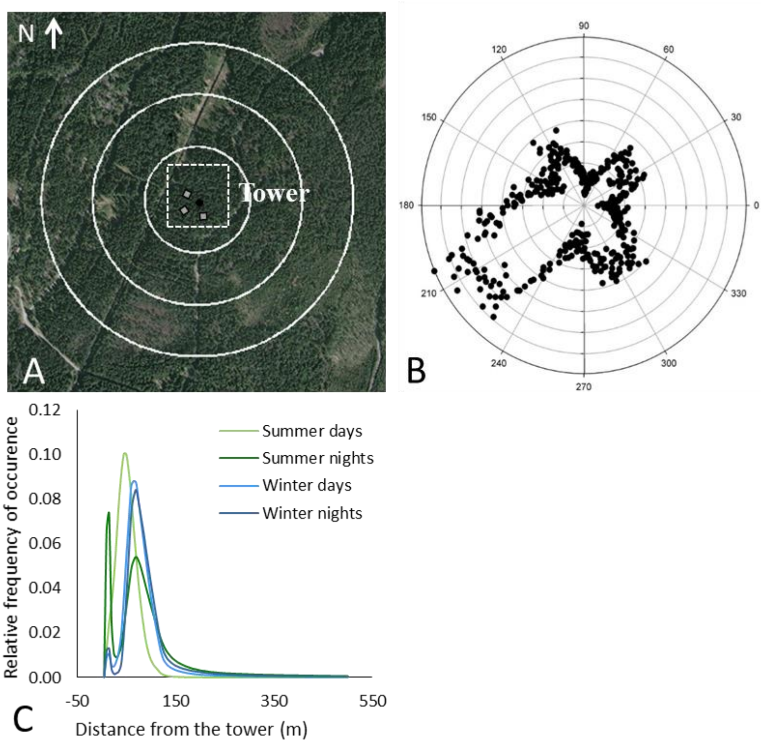

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Figure 1 (a) Position of the flux tower and of the three chamber measurement stations at the Skogaryd research catchment. The dotted line indicates the area of the 90 x 90 m plot in which trees have been sampled for quantification of biomass growth. 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°) and (c) Results of the footprint analysis.

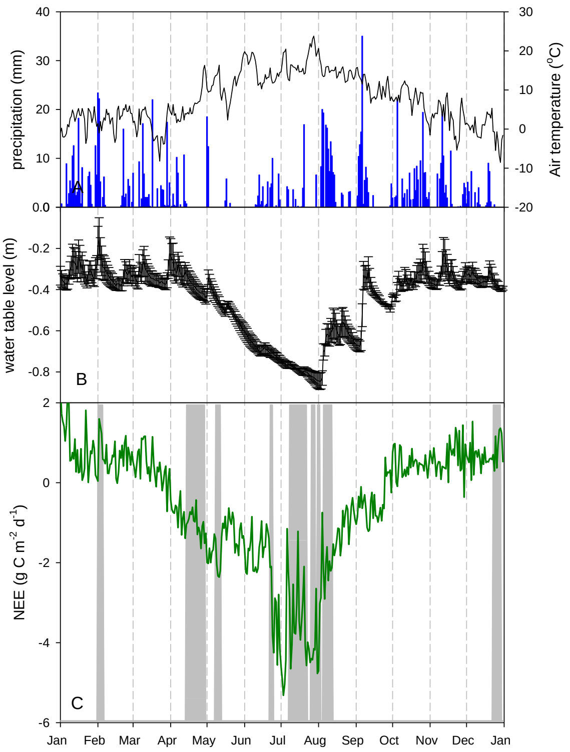
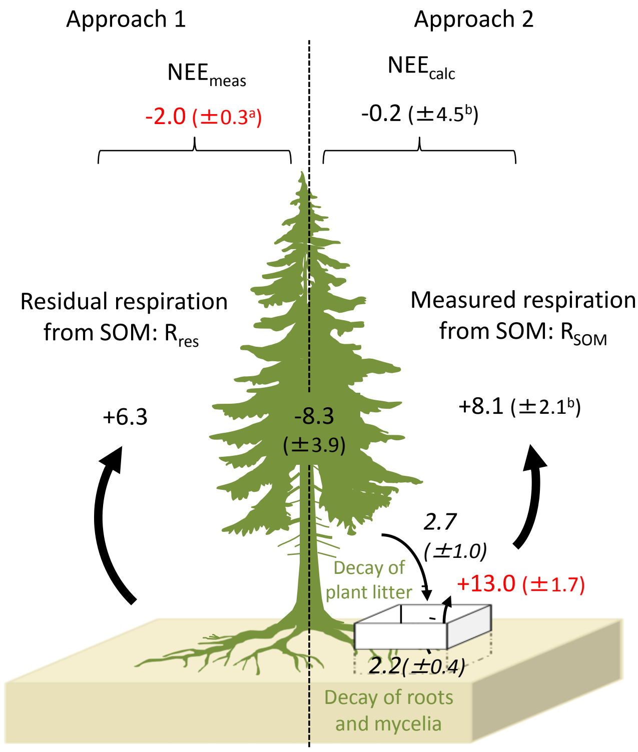


Figure 2 Daily precipitation (mm) from the Vänersborg SMHI weather station and average daily air temperature (˚C) (A), average water table level (B) and net ecosystem exchange (NEE: g C m^{-2} d^{-1}). The grey shadowed area indicates periods (>3 days) where data have been gap- filled (C).



[a] assuming 15% data uncertainty

[b] propagated error estimates are calculated by the Gaussian error propagation scheme assuming that components are independent

Figure 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_{meas} and the calculated tree growth. In approach 2 (right), the NEE is calculated as the residual flux NEE_{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. Fluxes determined as residuals are given in grey boxes. All fluxes in t C ha^{-1}yr^{-1}