

# Greenhouse gas dynamics of a well-drained afforested agricultural peatland

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Received 9 Dec. 2019, final version received 3 June. 2020, accepted 3 June. 2020

Ernfors M., Björk R.G., Noursratpour A., Rayner D., Weslien P. & Klemedtsson L. 2020: Greenhouse gas dynamics of a well-drained afforested agricultural peatland. *Boreal Env. Res.* 25: 65–89.

About a quarter of the global anthropogenic greenhouse gas emissions are attributable to agriculture, forestry and other land use. Few studies of afforested drained peatlands have measured exchanges of all three major greenhouse gases (GHG) at a given site, leading to uncertainty in estimated GHG budgets. Thus, we measured forest floor exchange of carbon dioxide (CO<sub>2</sub>), methane (CH<sub>4</sub>) and nitrous oxide (N<sub>2</sub>O) at a well-drained afforested peatland in southern Sweden. The CO<sub>2</sub> emissions (76 000 kg ha<sup>-1</sup> a<sup>-1</sup>; SE 6 000) were large compared with previous measurements at similar sites, which may have been partly due to a measurement technique that did not underestimate the flux. A net CH<sub>4</sub> uptake of 4.4 kg ha<sup>-1</sup> a<sup>-1</sup> (SE 0.41) and a net N<sub>2</sub>O emission of 2.7 kg ha<sup>-1</sup> a<sup>-1</sup> (SE 0.23) were found, which agreed well with published models relating fluxes to stand biomass (CH<sub>4</sub> models) or soil C:N ratio (N<sub>2</sub>O models).

## Introduction

Carbon dioxide (CO<sub>2</sub>), methane (CH<sub>4</sub>) and nitrous oxide (N<sub>2</sub>O) make up 98% of global anthropogenic greenhouse gas (GHG) emissions and about a quarter of these originate from land use (IPCC 2013). Particularly, agriculture and forestry on drained peatlands have a large impact on the anthropogenic greenhouse gas emissions (e.g., Kasimir Klemedtsson *et al.* 1997, Alm *et al.* 2007a, Minkkinen *et al.* 2008). In the boreal and temperate zones, approximately 15 Mha of drained peatlands are used for forestry (Paavilainen and Päivänen 1995) and approximately 14 Mha are used for agricultural purposes

(Armentano and Menges 1986). Peatlands in their natural state are long-term sinks for CO<sub>2</sub> (Gorham 1991, Clymo *et al.* 1998) and sources of CH<sub>4</sub> (Bartlett and Harris 1993, Nilsson *et al.* 2001), while N<sub>2</sub>O fluxes are close to zero (Martikainen *et al.* 1993, Regina *et al.* 1996). When peatlands are drained, the oxygen content of the peat increases, leading to increased CO<sub>2</sub> emissions (Silvola *et al.* 1996a, Trettin *et al.* 2006) and reduced or sometimes negative CH<sub>4</sub> emissions (Nykänen *et al.* 1998, Minkkinen and Laine 2006). Generally, if a peatland is drained for agricultural purposes, the net effect is a loss of carbon (C; Kasimir Klemedtsson *et al.* 1997, Maljanen *et al.* 2007). However, if the

drained peatland is used for forestry, it can be either a C sink or source, depending on whether or not the C emitted from the peat is balanced by an increase in tree stand biomass and/or an increased input of litter to the soil (Minkkinen *et al.* 2002, von Arnold 2004, Laiho 2006, Lohila *et al.* 2007, Meyer *et al.* 2013, Ojanen *et al.* 2014). In addition to affecting C fluxes, drainage can lead to N<sub>2</sub>O emissions if a peatland is sufficiently rich in nitrogen (N), i.e., if it has a low C:N ratio (Martikainen *et al.* 1993). There have been few studies taking all GHG emissions at a site into account, which increases the uncertainty of regional GHG budgets for drained peatlands (Maljanen *et al.*, 2010). Thus, more CO<sub>2</sub>, CH<sub>4</sub> and N<sub>2</sub>O flux data measured concurrently at a given site, paired with a better understanding of the main drivers of emissions, is clearly needed.

Peatlands that are used for agricultural purposes are generally chosen due to their high soil N content and, consequently, this land-use option can lead to significant release of N<sub>2</sub>O to the atmosphere (Kasimir-Klemedtsson *et al.* 1997, Maljanen *et al.* 2007). Many afforested peatlands have first been used for agricultural purposes before being planted with trees (Wall and Heiskanen 2003). Since these afforested former agricultural peatlands are likely to emit large amounts of GHG emissions, they are potential targets for mitigation measures and are key areas to consider when scaling up greenhouse gas fluxes from forest land to regional or national levels (Ernfors *et al.* 2007). Studies in Finland have shown that N<sub>2</sub>O emissions from drained afforested former agricultural peatlands are close to those from organic peatlands in active cultivation and greater than those from organic soils drained for forestry (Martikainen *et al.* 1993, Alm *et al.* 2007a, Maljanen *et al.* 2007, Mäkiranta *et al.* 2007, Maljanen *et al.* 2012). However, for Sweden there are few published studies on greenhouse gas exchange at drained afforested peatlands that have previously been used for agriculture. Klemedtsson *et al.* (2010) and Rütting *et al.* (2014) measured greenhouse gas emissions in a wood ash fertilization experiment at the drained and afforested agricultural peat site in the Skogaryd research catchment in southern Sweden over two and four years, respectively. Their results from untreated control

plots indicated that N<sub>2</sub>O and CO<sub>2</sub> emissions were moderate compared with those from drained peatlands under agricultural use (Kasimir-Klemedtsson *et al.* 1997, Maljanen *et al.* 2012) and that there was a relatively large uptake of CH<sub>4</sub> compared with cultivated (Maljanen *et al.* 2007) and afforested agricultural (Maljanen *et al.* 2003, Mäkiranta *et al.* 2007) organic soils in Finland, as well as compared with cultivated or forestry drained peatlands in Sweden (Kasimir-Klemedtsson *et al.* 1997, von Arnold *et al.* 2005a-c, Weslien *et al.* 2009). Meyer *et al.* (2013) measured heterotrophic respiration in trenched plots at the Skogaryd drained peat site in 2008 and obtained an annual flux value higher than those for total forest floor respiration measured at the same site by Klemedtsson *et al.* (2010) and Rütting *et al.* (2014). This emphasizes the need for further quantification and explanation of GHG fluxes from afforested peat soils.

Several authors have constructed models to try to predict CH<sub>4</sub> and N<sub>2</sub>O fluxes at drained forested peatlands, based on easily measured tree stand or soil parameters (Klemedtsson *et al.* 2005, Ernfors *et al.* 2007, Minkkinen *et al.* 2007 and Ojanen *et al.* 2010). Minkkinen *et al.* (2007) proposed a model ( $R^2 = 0.58$ ), based on measurements in 26 plots, linking annual CH<sub>4</sub> emissions from drained peatlands to stand volume. They argued that since tree volume reflects both drainage age and the nutrient status of the peat, it forms a causal link to CH<sub>4</sub> fluxes. Drainage age and nutrient status of the peat are factors influencing the main determinants of CH<sub>4</sub> fluxes: groundwater table and the composition of the ground vegetation (Whalen *et al.* 2005). However, Ojanen *et al.* (2010) found a weaker relationship ( $R^2 = 0.22$ ) between stand volume and CH<sub>4</sub> fluxes. To predict annual N<sub>2</sub>O emissions, Klemedtsson *et al.* (2005) constructed a model ( $R^2 = 0.96$ ), based on measurements on 12 sites, where they linked annual N<sub>2</sub>O emissions from drained peatlands to the C:N ratio of the top soil. Ernfors *et al.* (2007) modified this model in order to calculate a confidence band around it ( $R^2 = 0.87$  for the new model). Ojanen *et al.* (2010) constructed three models based on measurements of N<sub>2</sub>O emissions and top soil C:N ratios of drained peatlands. First, one model

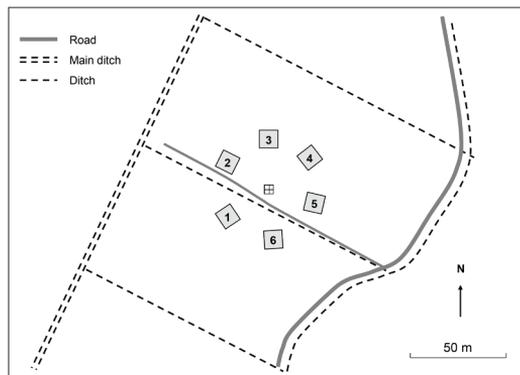
using their own data ( $R^2 = 0.50$ ); second, one using their own data together with data from other sites drained for forestry ( $R^2 = 0.35$ ); and third, one also including the data used in Klemetsson *et al.* (2005) and Ernfors *et al.* (2007;  $R^2 = 0.24$ ). These existing models need to be evaluated against new empirical data in order to improve the upscaling of fluxes to larger regions.

The objective of our study was to quantify the net forest floor exchange of  $\text{CO}_2$ ,  $\text{CH}_4$  and  $\text{N}_2\text{O}$  at a site representative of Swedish afforested drained agricultural peatlands and to relate these GHG fluxes to abiotic variables, e.g., by investigating how well they could be predicted using the models by Minkkinen *et al.* (2007), Ojanen *et al.* (2010), Klemetsson *et al.* (2005) and Ernfors *et al.* (2007). We hypothesised that: (i) the forest floor  $\text{CO}_2$  emissions at our site would be larger than those previously measured by von Arnold *et al.* (2005a, 2005b, 2005c) at poorly drained forested Swedish peatlands, since a lower groundwater table promotes increased  $\text{CO}_2$  fluxes (Silvola *et al.* 1996a, Mäkiranta *et al.* 2009); (ii) our site would be a net sink for  $\text{CH}_4$ , due to its low groundwater table and large tree stand, in agreement with the models presented by Minkkinen *et al.* (2007) and Ojanen *et al.* (2010); and (iii) the annual  $\text{N}_2\text{O}$  flux at our site could be predicted from the C:N ratio of the soil using the relationship described by Ernfors *et al.* (2007).

## Material and methods

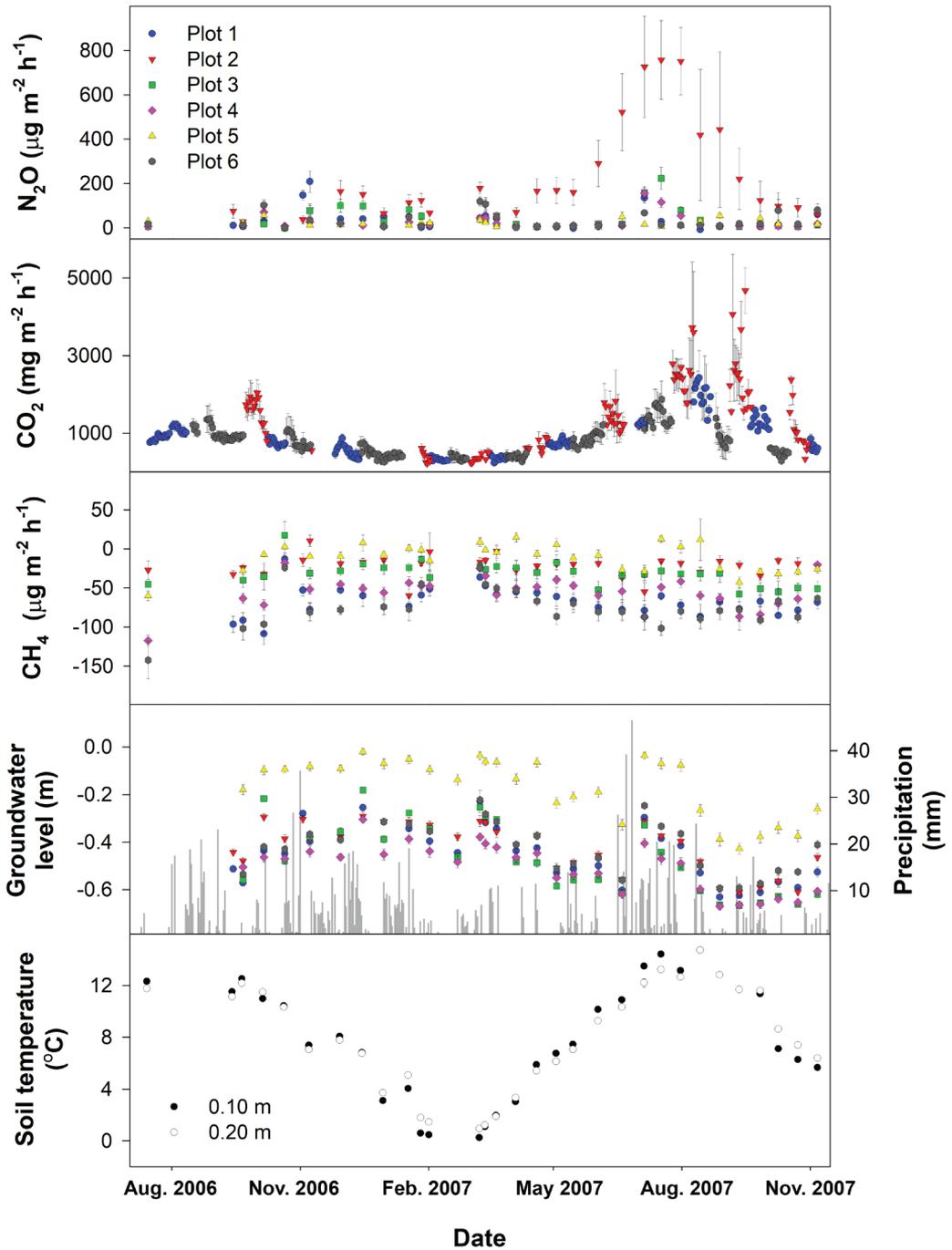
### Site

The study was carried out at the Skogaryd research catchment in southwest Sweden ( $58^\circ23'N$ ,  $12^\circ09'E$ ; 60 metres above sea level). The long-term (1961–1990) mean annual temperature and precipitation were  $6.4^\circ\text{C}$  and 709 mm, respectively, at the Vänersborg weather station, 12 km from the site (Alexandersson and Eggertsson Karlström 2001). The Skogaryd drained peat site is located at a well-drained former fen, formed within a valley. The peat is between one and several meters deep and mineral particles are incorporated into the peat in



**Fig 1.** Map of the drained peat site in the Skogaryd research catchment. The small square in the middle represents a micrometeorological tower and the numbered squares represent the plots where the measurements of gas fluxes and most abiotic parameters were conducted.

varying amounts. Below the peat layer, there is a clay sediment of marine origin. The mineral component of the peat is likely to originate either from flood events or from additions of mineral soil for fertilization purposes. The site was drained in the 1870s and used for agriculture until it was planted with Norway spruce (*Picea abies*) in the 1950s. The height of the tree stand and the stand volume were approximately 22 m and  $400 \text{ m}^3 \text{ ha}^{-1}$ , respectively, at the start of the study (2006). The ground vegetation was sparse or absent in the denser parts of the stand and dominated by bryophytes in the more open parts. The bottom layer consisted mainly of *Plagiomnium affine*, *Dicranum Majus*, *Polytrichum formosum*, *Pleurozium schreberi* and *Hylocomium splendens*, while *Vaccinium myrtillus*, *Luzula pilosa*, *Oxalis acetosella* and *Juncus conglomeratus* were common in the field layer (Björk *et al.* 2010). We measured gas fluxes between the soil and the atmosphere, along with abiotic soil properties, at six plots. The plots were arranged in a circle ( $\varnothing 50 \text{ m}$ ) in the estimated main footprint area of a micrometeorological tower (Fig. 1), to allow for subsequent comparisons between chamber and micrometeorological data. Two of the plots (plots 2 and 5) had properties that set them apart from the others and they may not have been representative of the site as a whole. These plots were not relocated since the area around the tower had to be cov-



**Fig. 2.** Daily average gas fluxes, precipitation and manually measured groundwater levels and manually measured soil temperatures at the drained peat site in the Skogaryd research catchment, over the period from 15 Jul. 2006 to 6 Nov. 2007.

ered in all directions. In plot 2 (Fig. 1), there was a natural deviation in soil texture. Soil profiles in the plots showed that there was a sediment layer at a depth of 25–30 cm in parts of plot 2, which was not present in the other plots. Plot 5 (Fig. 1) was located on a track that had been used by forest machinery during a forest thinning operation. The forest machines had created a depression in the soil, leading to wet conditions in the whole plot (Fig. 2). There were also no trees on the track, which led to more favorable light conditions and more field and bottom layer vegetation than in the other plots. The vegetation in plot 5 included species characteristic of wet soil, such as *Sphagnum spp.*

### Gas flux measurements

We installed six stainless steel collars at each plot and left them in place during the whole experiment. At each plot, we distributed the collars in a random pattern, within a 10 m × 10 m square. Every collar had a channel at the top, into which a chamber could be placed. Water or a rubber gasket in the channel ensured an air-tight seal between chamber and collar. The collars measured 44 cm × 44 cm on the inside and had a horizontal flange extending 7 cm outwards at the lower end, both to allow the collar to be more securely fixed onto the soil surface, thus minimizing the risk of leaks and to decrease leakage through the soil pore space by increasing the path of diffusion between the chamber and the ambient air. We assumed that the diffusion flux from the soil divided at the middle of the flange and, accordingly, the effective uptake area of each chamber was 0.26 m<sup>2</sup>. We aimed at installing the collars with the flange just beneath the litter layer to avoid cutting any fine roots or mycorrhizae. However, due to the uneven soil surface, it was often necessary to install parts of the collars a few centimetres deeper, to a maximum of 5 cm into the peat. More than half the circumference of each collar always remained on top of the peat layer. We measured gas fluxes between 14 Jul. 2006 and 6 Nov. 2007. Only two of the collars in plots 1, 2 and 6 were used for the entire period, as four of the collars were used for another study from 6 Aug. 2007 onwards.



**Fig 3.** One of six automatic chambers used to measure CO<sub>2</sub> fluxes.

### Automatic chamber measurements of CO<sub>2</sub>

For the measurements of CO<sub>2</sub> fluxes, we used six automatic closed chambers. The chambers were 1.25 m high and made out of transparent polycarbonate sheets on aluminium frames (Fig. 3). The mean combined volume of a chamber and its collar, calculated from the soil surface, was 0.24 m<sup>3</sup>. Each chamber had a lid at the top, attached by hinges on one side and opened with an electric actuator (Linear Actuator LA12; LINAK A/S, Nordborg, Denmark). The lid was in a vertical position when opened. A spiral shaped vent, made from a PVC pipe (inner  $\varnothing$  4 mm), was installed through the lid of each chamber. An air intake was positioned in the middle of the chamber, approximately 40 cm from the soil surface. A ring of tubing around the bottom of the chamber, approximately 10 cm

from the soil surface, with multiple small holes, served as an air outlet manifold. A  $\varnothing$  80 mm fan (Jamicon, New Taipei City, Taiwan;  $32.1 \text{ L s}^{-1}$ ) was positioned horizontally, at approximately 50 cm from the soil surface, and served to ventilate the chamber between measurements. Two  $\varnothing$  50 mm fans (Jamicon, New Taipei City, Taiwan;  $7.6 \text{ L s}^{-1}$ ) were positioned at an angle at the sides of the chamber, approximately 20 cm from the soil surface, and served to mix the air within the chamber during a measurement. The chambers were connected to a logger system (CR10X logger, AM16/32A multiplexer, SDM-CD16 relay control port module; Campbell Scientific Inc., Logan UT, USA) which prompted a measurement cycle every 20 minutes, recording values from two chambers at a time, resulting in one gas measurement per chamber per hour. During a measurement, the lid closed and air was pumped from the chamber, through an air filter, to an infrared gas analyser (PP Systems SBA-4 OEM  $\text{CO}_2$  Analyzer) and back to the chamber. The gas analyser measured the  $\text{CO}_2$  concentration in the chamber air every 30 seconds, for 15 minutes, before the lid was opened again. The logger stored the  $\text{CO}_2$  concentration values and time points for the measurements. We moved the chambers approximately every fortnight, circulating between plots 1, 2 and 6 (Fig. 1). Since the chambers were transparent, we removed all vegetation from inside the collars in these plots, to eliminate  $\text{CO}_2$  uptake by photosynthesis.

### Manual chamber measurements of $\text{CH}_4$ and $\text{N}_2\text{O}$

For measurements of  $\text{CH}_4$  and  $\text{N}_2\text{O}$ , we used manual dark static chambers made of stainless steel, from which gas samples could be collected through a butyl rubber membrane. We used the same collars for the manual chambers as for the automatic chambers and we joined the chamber and collar in the same way, using water or rubber gaskets to provide a gas tight seal. We collected gas samples in 22 ml glass vials, sealed with butyl rubber membranes, by circulating air for 30 s between the chamber headspace and a vial, through two  $\varnothing$  0.7 mm needles. We collected samples at 4, 8, 16 and 32 min after placing the

chamber on the collar and analysed them using a Varian 3800 gas chromatograph, as described by Klemetsson *et al.* (1997). The mean volume of the combination of chamber and collar, calculated from the soil surface, was  $0.028 \text{ m}^3$ . We measured gas fluxes using the manual chambers on 25 occasions during the period from 14 Jul. 2006 to 31 Jul. 2007 (approximately fortnightly). After that, we performed seven additional measurements during the period from 1 Aug. to 6 Nov. 2007, to track the peak in the  $\text{N}_2\text{O}$  flux in plot 2 (Fig. 2).

### Abiotic measurements

In connection to the measurements of  $\text{CH}_4$  and  $\text{N}_2\text{O}$ , we manually measured soil temperature and groundwater level. We used a steel thermoelement sensor (TPK-03, Tecpel, Taipei, Taiwan) connected to a handheld thermometer (CIE 307, Tecpel, Taipei, Taiwan) to measure soil temperature at 10 and 20 cm depth, approximately 30 cm from the collars. During the period from 14 Jul. 2006 to 29 Nov. 2006, we measured soil temperature in three of the plots and afterwards in all plots. To determine the depth of the groundwater table, we used a plumb line, in perforated tubes inserted to a depth of 1.5 m in the soil, approximately 50 cm from each collar.

To determine pH and total C and N contents, we collected soil samples at depths of 0–5 cm and 5–20 cm, approximately 50 cm from each collar in Feb. 2008. We measured pH in a 1:10 (by fresh weight) deionized water suspension and total C and N using an elemental analyser (Model: EA 1108 CHNS-O; Fisons Instruments S.p.A., Rodano, Italy).

From 14 Aug. 2006, a logger system (CR10X logger, AM16/32A multiplexer; Campbell Scientific Inc., Logan UT, USA) recorded soil moisture, soil temperature and air temperature at the site, every 30 min. The logger system obtained the soil moisture measurements from CS616 Water Content Reflectometers (Campbell Scientific Inc., Logan UT, USA), which were inserted horizontally into the soil at depths of 5, 15, 30 and 60 cm, in two soil profiles (a and b) in each of the plots 1, 2 and 6. We carried out a site-specific calibration, to relate the CS616

Water Content Reflectometer signal to water content, since standard calibration coefficients do not apply to peat soils (Campbell Scientific Inc., 2006). The calibration function that we used (see Eq. 1), where  $y$  = soil water content (g water  $\times$  g dry weight<sup>-1</sup>) and  $x$  = CS616 output value, was based on 16 pairs of gravimetric water content and CS616 output values, from field and laboratory measurements, with  $R^2 = 0.66$ .

$$y = 11.139x^2 - 6.256x + 2.794 \quad (1)$$

The logger recorded soil temperature measurements from Campbell 107 Temperature Probes (Campbell Scientific Inc., Logan UT, USA) and copper-constantan (Type T) thermocouples, installed in the same soil profiles and at the same depths as the soil moisture probes. Air temperature was measured 2 m above ground, using a Campbell 107 Temperature Probe (Campbell Scientific Inc., Logan UT, USA) equipped with a naturally-aspirated radiation shield (UT12VA 12-Plate Radiation Shield; Campbell Scientific Inc., Logan UT, USA). Wind speed was measured at 27 m height in the micrometeorological tower (Fig. 1) with a Gill 3 sonic anemometer (Solent, UK).

## Calculations and statistics

We calculated the flux of each gas from the linear part of the regression slope of the chamber gas concentration values, plotted against time. For the manual gas measurements, we regarded all slope values with  $R^2 > 0.90$  as correct and accepted slopes with  $R^2 < 0.90$  if the change in gas concentration within the chamber, during the measurement, was less than 20%. When fluxes are small, the  $R^2$ -values are inherently low and should not be used to disqualify a measurement (Alm *et al.* 2007b). Minor changes in the gas concentration in the chamber, due to disturbances in the field, have a relatively larger impact on the linearity of the regression slope at low fluxes and excluding the small fluxes due to low  $R^2$ -values would lead to an overestimate of the total annual fluxes.

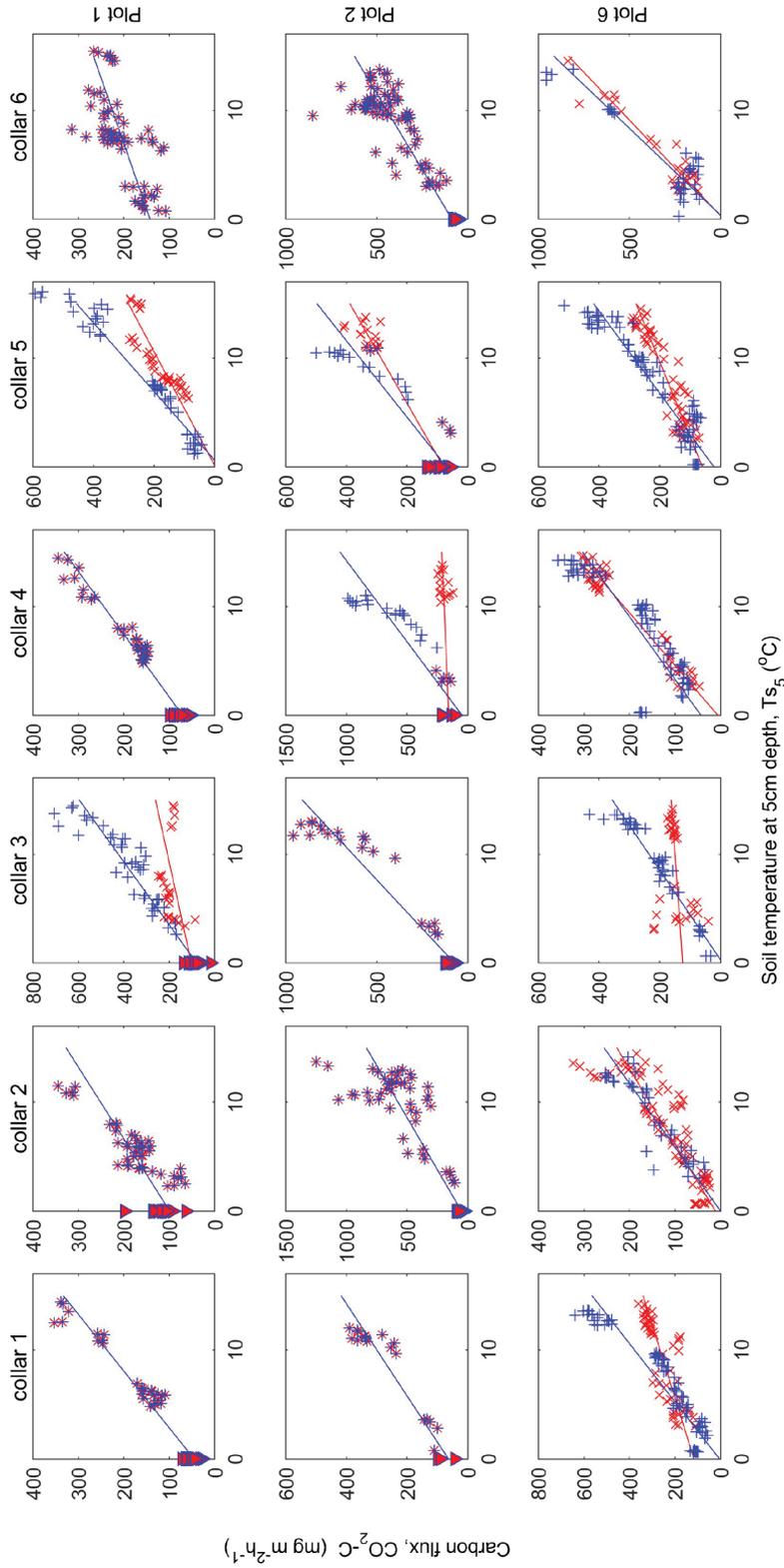
For CH<sub>4</sub> and N<sub>2</sub>O, we calculated annual flux values for each collar using temporal integra-

tion of the emission values, for the period from 1 Aug. 2006 to 31 Jul. 2007. For N<sub>2</sub>O, we also calculated annual fluxes for the period from 7 Nov. 2006 to 6 Nov. 2007, then including the additional measurements that were made to study the N<sub>2</sub>O flux peak at plot 2 (Fig. 2). For this calculation, only the two collars in each of plots 1, 2 and 6 that were present for the entire period were used. When calculating the site average, fluxes were first averaged for each plot, to ensure that all plots influenced the average equally.

Two statistical methods were used to estimate annual CO<sub>2</sub> fluxes from the measured values. In the first method, we gapfilled and then integrated the data over the period from 1 Aug. 2006 to 31 Jul. 2007, to obtain annual flux values for each collar. Multiple regression of the ln-transformed CO<sub>2</sub> flux against automatically measured soil temperature, soil moisture and groundwater table data showed that soil temperature at 5 cm depth ( $T_{s_5}$ ) and soil water content at 30 cm depth ( $w_{30}$ ) were the variables that explained most of the variation in the CO<sub>2</sub> flux, for the whole data set ( $R^2_{\text{adj}} T_{s_5} = 0.51$ ;  $R^2_{\text{adj}} T_{s_5} + w_{30} = 0.55$ ; residuals were normal and homoscedastic,  $F$  and  $p$  values were not valid due to autocorrelation). We therefore gapfilled the data using these two variables (Eq. 2), with  $a$ ,  $b$  and  $c$  constants determined separately for each collar. When  $T_{s_5}$  and/or  $w_{30}$  data were missing, we gapfilled using linear interpolation or extrapolation.

$$\ln \text{CO}_2 = a + b(T_{s_5}) + c(w_{30}) \quad (2)$$

In the second method, annual fluxes were estimated based solely on the observed relationship between daily-average CO<sub>2</sub> flux and  $T_{s_5}$ . The relationship between daily CO<sub>2</sub> flux and  $T_{s_5}$  is clearly not stationary — for many collars, there are two approximate relationships between CO<sub>2</sub> flux and  $T_{s_5}$  (Fig. 4). In most of these cases, the relationship is different for 2006 and 2007, although for several collars the bifurcation in the relationship does not follow the 2006/2007 division. The bifurcation could not be accounted for by introducing soil water content as an independent variable. In short, for many collars, it was not possible to predict which relationship should be preferred.



**Fig. 4.** Daily average carbon flux as a function of soil temperature at 5 cm depth, for all collars in the study. The blue "+" indicate observations used in the "high" regression model (blue lines); red "x" indicate observations used in the "low" model (red line); overlaid "+" and "x" indicate observations used in both models; triangles indicate observations where  $T_{s_5}$  values less than  $0^{\circ}\text{C}$  were treated as  $0^{\circ}\text{C}$  in the regressions, used in both models. Observation period covers 1 Aug. 2006 to 06 Nov. 2007.

Thus, in the second method, we chose to account for this uncertainty by creating two linear regressions for those collars where the data indicated two possible relationships (Fig. 4), a "high" model (mainly data from 2007) and a "low" model (mainly data from 2006). For collars without two clear relationships, we created "high" and "low" models with the same data so as to evenly weight the collars when calculating annual averages. We assumed CO<sub>2</sub> flux to be independent of temperature below freezing, thus T<sub>s</sub> values less than 0°C were treated as 0°C in the regressions. The mean-absolute-errors of the "low" linear fits relative to the observations are typically 92–163 mg CO<sub>2</sub> m<sup>-2</sup> h<sup>-1</sup>, implying typical errors of 12–18% (range given as the 25th and 75th percentiles). The "high" fits were somewhat poorer, with mean-absolute-errors ranging from 106–250 mg CO<sub>2</sub> m<sup>-2</sup> h<sup>-1</sup>, implying typical errors of 13–22%.

Both regression fits were then used with the entire T<sub>s</sub> record to create a modelled "high" and "low" CO<sub>2</sub> flux time-series for each collar covering the period from 1 Aug. 2006 to 6 Nov. 2007. Gaps in the modelled CO<sub>2</sub> flux time-series (i.e., for days without a valid T<sub>s</sub>, representing 2.6% of all days) were filled using robust LOWESS smoothing. The time-series were then averaged to day-of-year and then to annual averages.

The final estimate of the annual CO<sub>2</sub> flux for each plot for our second method was then calculated as the mean of both the "high" and "low" averages for all collars in the plot. The final estimate of the annual CO<sub>2</sub> flux for the site was calculated as the weighted average across all collars, with weights inversely-proportional to the number of collars in each plot.

Annual CH<sub>4</sub> and N<sub>2</sub>O values for the site were also calculated using the models presented by Minkkinen *et al.* (2007), Ojanen *et al.* (2010), Klemedtsson *et al.* (2005) and Ernfors *et al.* (2007), based on abiotic data measured at the Skogaryd drained peat site. Tree stand volume at the Skogaryd drained peat site, which was estimated from clinometer measurements of tree heights and caliper measurements of stem diameters, was used with the Minkkinen *et al.* (2007) and Ojanen *et al.* (2010) equations for CH<sub>4</sub>. The highest tree stand volume was close to 300 m<sup>3</sup> ha<sup>-1</sup> in both the Minkkinen *et al.* (2007)

and Ojanen *et al.* (2010) studies. The stand volume at the Skogaryd drained peat site was estimated to 400 m<sup>3</sup> ha<sup>-1</sup> and an extrapolation was needed to obtain CH<sub>4</sub> flux values from the models. C:N ratios of the 0–5 cm and 0–20 cm soil layers at the Skogaryd drained peat site were used with the Ernfors *et al.* (2007), Ojanen *et al.* (2010) and Klemedtsson *et al.* (2005) equations for N<sub>2</sub>O.

For the analyses of spatial variation in the data, we calculated weighted annual averages for groundwater level, soil moisture and automatically measured soil temperature, for each collar. For manually-measured soil temperature, we calculated weighted averages for the 2007 growing season (May–Oct.), since annual data was not available for all collars.

We used one-way ANOVA to analyse for differences in gas fluxes and abiotic variables between plots. For post-hoc comparisons, we used the Tukey HSD test when the data were homoscedastic and the Games-Howell test when the data were heteroscedastic and could not be improved by transformations. To evaluate our CO<sub>2</sub> flux measurement method, we calculated Pearson's correlation coefficients for wind speed versus CO<sub>2</sub> flux. The statistical analyses were performed using SPSS (SPSS 14.0, SPSS Inc., Chicago, IL, USA) and MATLAB (The MathWorks, Inc. Massachusetts, USA). We used a statistical significance level ( $\alpha$ ) of 0.05 for all tests.

## Results

### CO<sub>2</sub>

The total annual forest floor CO<sub>2</sub> emission, over the main measurement period (1 Aug. 2006 to 31 Jul. 2007) was 72 000 kg CO<sub>2</sub> ha<sup>-1</sup> (SE 5 900 kg CO<sub>2</sub> ha<sup>-1</sup>) using the first method and 76 000 kg CO<sub>2</sub> ha<sup>-1</sup> (SE 6 000 kg CO<sub>2</sub> ha<sup>-1</sup>) using the second method (Table 1). The two methods for gapfilling thus yielded similar annual values. We considered the second method as the more reliable since this method took into account the differences between years in the relation between daily CO<sub>2</sub> flux and T<sub>s</sub>. We therefore used the results from the second method in the Discussion and in Table 1. The

CO<sub>2</sub> flux exhibited a clear seasonal pattern with lower fluxes in the winter and spring and higher fluxes in the summer and autumn (Fig. 2). The annual CO<sub>2</sub> flux differed between plots (ANOVA:  $F_2 = 3.7$ ,  $p = 0.049$ ), with fluxes at plot 2 significantly larger than for plot 1 (Games-Howell:  $p = 0.024$ ), while plot 6 did not differ from plots 1 or 2 with respect to CO<sub>2</sub> emissions. For plots 1, 2 and 6 together, there was no correlation between hourly CO<sub>2</sub> flux and wind speed values ( $r = -0.004$ ,  $p = 0.72$ ,  $n = 7761$ ). When analysed separately, there was a weak correlation between hourly CO<sub>2</sub> flux and wind speed ( $r = 0.087$ ,  $p = 0.000$ ,  $n = 2041$ ) at plot 2, but no statistically significant differences at plots 1 and 6.

## CH<sub>4</sub>

The total annual uptake of CH<sub>4</sub> at the site (1 Aug. 2006 to 31 Jul. 2007) was

**Table 1.** Annual gas flux values at the drained peat site in the Skogaryd research catchment, during the period of 1 Aug. 2006 to 31 Jul. 2007.

Gas	Plot	Annual flux (kg ha <sup>-1</sup> )
CH <sub>4</sub>	1	-6.0 ± 0.6
	2	-2.0 ± 0.3
	3	-2.5 ± 0.6
	4	-4.7 ± 0.4
	5	-0.9 ± 0.1
	6	-6.7 ± 0.6
	All	-3.8 ± 0.4
	All except 5	-4.4 ± 0.4
CO <sub>2</sub>	1	60 000 ± 5 000
	2	98 000 ± 10 000
	6	71 000 ± 10 000
	1, 2 and 6	76 000 ± 6 000
N <sub>2</sub> O	1	3.3 ± 0.5
	2	16 ± 4
	3	3.9 ± 0.7
	4	2.5 ± 0.3
	5	1.6 ± 0.1
	6	2.4 ± 0.2
	All	5.0 ± 1.1
	All except 2	2.7 ± 0.2

3.8 kg CH<sub>4</sub> ha<sup>-1</sup> (SE 0.40) calculated for all plots and 4.4 kg CH<sub>4</sub> ha<sup>-1</sup> (SE 0.41) when we excluded plot 5 (see Discussion section; Table 1). The models by Minkkinen *et al.* (2007) and Ojanen *et al.* (2010) predicted CH<sub>4</sub> uptakes of 5.9 kg ha<sup>-1</sup> a<sup>-1</sup> and 4.2 kg CH<sub>4</sub> ha<sup>-1</sup>, respectively, when extrapolated to the estimated tree stand volume of 400 m<sup>3</sup> at the Skogaryd drained peat site. In plot 5, the annual CH<sub>4</sub> uptake was significantly lower (one-way ANOVA:  $F_5 = 24.7$ ,  $p < 0.001$ ) than in plots 1, 4 and 6 (Games-Howell:  $p = 0.002$ ,  $p = 0.001$  and  $p = 0.001$ , respectively). When we excluded plot 5 (see Discussion section), the CH<sub>4</sub> uptake was significantly higher (one-way ANOVA:  $F_4 = 16.4$ ,  $p < 0.001$ ) in plots 1, 4 and 6 than in plots 2 and 3 (Tukey HSD: 1 and 6 vs. 2 and 3, all  $p < 0.001$ ; 4 vs. 2,  $p = 0.009$ ; 4 vs. 3,  $p = 0.044$ ). Despite statistically significant differences between plots, both the spatial and the temporal variations in CH<sub>4</sub> fluxes at the site were small (Table 1, Fig. 2).

## N<sub>2</sub>O

The total annual N<sub>2</sub>O flux at the site, calculated over the period from 1 Aug. 2006 to 31 Jul. 2007, was 5.0 kg N<sub>2</sub>O ha<sup>-1</sup> (SE 1.10) calculated for all plots and 2.7 kg N<sub>2</sub>O ha<sup>-1</sup> (SE 0.23) when we excluded plot 2 (see Discussion section; Table 1). Annual fluxes calculated over the period from 7 Nov. 2006 to 6 Nov. 2007, including the whole peak in plot 2 (Fig. 2), were 5.7 kg N<sub>2</sub>O ha<sup>-1</sup> (SE 2.9) when we included all plots and 2.9 kg N<sub>2</sub>O ha<sup>-1</sup> (SE 0.47) when we excluded plot 2.

The five N<sub>2</sub>O emission values derived for the Skogaryd drained peat site using the models by (1) Ernfors *et al.* (2007); (2) Klemetsson *et al.* (2005); (3) Ojanen *et al.* (2010) — based on their own data; (4) Ojanen *et al.* 2010 — based on their own data and other data from forestry drained peatlands; and (5) Ojanen *et al.* (2010) — based on their own data and other data, including afforested agricultural sites, were 3.0, 1.4, 0.8, 1.7 and 1.0 kg ha<sup>-1</sup> a<sup>-1</sup>, respectively, when using the C:N ratios of the 0–5 cm (C/N = 21) and 1.9, 0.4, 0.7, 1.2 and 0.3 kg ha<sup>-1</sup> a<sup>-1</sup>, respectively, when using the C:N ratios of the 0–20 cm (C/N = 24) soil layers.

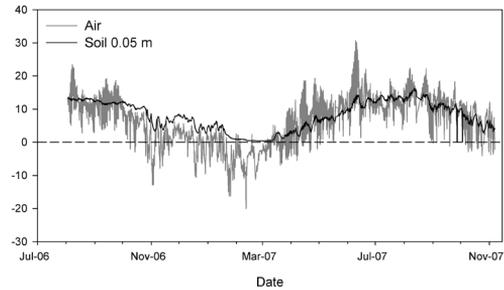
The N<sub>2</sub>O emissions at the Skogaryd drained peat site varied considerably, both in space and time. Two peaks in the N<sub>2</sub>O emissions could be

discerned in all six plots: one in connection with the main spring thaw in Mar. 2007 (Figs. 2 and 5) and one in connection with heavy summer rains in Jul. 2007 (Fig. 2). In addition, there was a large N<sub>2</sub>O peak in plot 2 from Aug. to Nov. 2007, which was not present in the other plots.

The large late summer and early autumn N<sub>2</sub>O emission peak caused the average annual N<sub>2</sub>O flux from plot 2 to be six times higher than the mean of the other five plots (Table 1). The annual ln-transformed N<sub>2</sub>O flux at plot 2 was significantly higher (one-way ANOVA:  $F_5 = 18.6$ ,  $p < 0.001$ ) than that at plots 1, 3, 4, 5 and 6 (Games-Howell:  $p = 0.017$ ,  $p = 0.034$ ,  $p = 0.009$ ,  $p = 0.004$  and  $p = 0.009$ , respectively). The annual ln-transformed N<sub>2</sub>O flux at plot 5 was also significantly lower than at plot 6 (Games-Howell:  $p = 0.025$ ). When we excluded plot 2 from the analysis of the spatial variation in annual log-transformed N<sub>2</sub>O emissions, the emissions from plot 5 were significantly lower (one-way ANOVA:  $F_4 = 4.9$ ,  $p = 0.005$ ) than those from plots 3 and 6 (Games-Howell:  $p = 0.045$  and  $p = 0.018$ , respectively), and tended to be lower than from plot 4 (Games-Howell:  $p = 0.052$ ). Generally, the spatial variation in the N<sub>2</sub>O emissions was small when we excluded plot 2 (Table 1).

## Soil conditions

Soil temperature at the Skogaryd drained peat site varied between approximately 0°C and 15°C, at depths of both 10 and 20 cm (Fig. 2). At 10 cm depth, the average ln-transformed soil tempera-



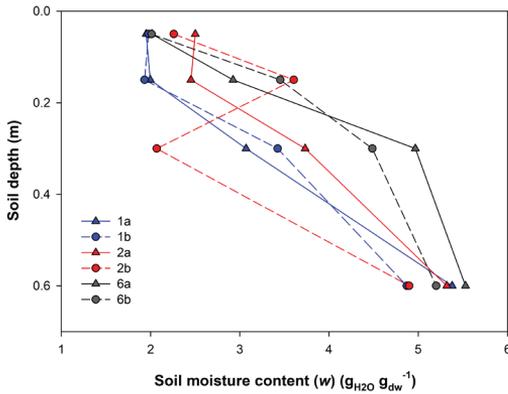
**Fig 5.** Automatically-measured air temperature (grey line) and soil temperature (black line) at a depth of 5 cm, at the drained peat site in the Skogaryd research catchment, over the period from 14 Aug. 2006 to 6 Nov. 2007. The dashed line indicates a temperature of 0°C.

ture during the growing season (May–Oct.) was significantly higher (one-way ANOVA:  $F_5 = 5.788$ ,  $p = 0.001$ ) at plot 5 compared with plots 2 and 6 (Tukey HSD:  $p = 0.002$  and  $p = 0.001$ , respectively). Similarly, at 20 cm depth, the average ln-transformed soil temperature during the growing season was significantly higher (one-way ANOVA:  $F_5 = 6.404$ ,  $p < 0.001$ ) at plot 5 compared with plots 2 and 6 (Tukey HSD:  $p = 0.006$  and  $p < 0.001$ , respectively) but also significantly higher in plots 3 and 4, compared with plot 6 (Tukey HSD:  $p < 0.025$  and  $p < 0.030$ , respectively). Despite statistically significant differences between plots, the spatial variation in growing season soil temperatures at the site was small (Table 2).

The groundwater level at the site varied between approximately 0 cm and 65 cm (Fig. 2). The mean annual groundwater table in plot 5 dif-

**Table 2.** Mean values of pH, C:N ratio, groundwater table and soil temperature at the drained peat site in the Skogaryd research catchment, with standard errors. The average groundwater table for the site is presented including/excluding plot 5. Soil temperature at 10 cm is the average from Aug. 2006–Jul. 2007; and at 20 cm, the average is from May–Oct. 2007.

Plot	pH		C/N		Groundwater table (m)	Soil temperature (°C)	
	0–5 cm	5–20 cm	0–5 cm	5–20 cm		10 cm	20 cm
1	4.8 ± 0.03	4.6 ± 0.06	21.2 ± 1.4	24.3 ± 1.1	0.43 ± 0.01	11.0 ± 0.06	10.8 ± 0.07
2	4.5 ± 0.07	4.2 ± 0.07	23.0 ± 1.1	23.8 ± 1.7	0.40 ± 0.01	10.8 ± 0.05	10.7 ± 0.04
3	4.5 ± 0.06	4.2 ± 0.06	22.3 ± 0.8	23.8 ± 1.5	0.43 ± 0.01	11.0 ± 0.01	10.8 ± 0.02
4	4.4 ± 0.06	4.2 ± 0.03	21.3 ± 1.1	21.0 ± 0.7	0.47 ± 0.02	11.0 ± 0.03	10.8 ± 0.04
5	4.5 ± 0.03	4.6 ± 0.04	19.7 ± 1.2	32.2 ± 2.3	0.12 ± 0.02	11.1 ± 0.09	10.9 ± 0.08
6	4.7 ± 0.06	4.6 ± 0.05	18.2 ± 0.7	25.8 ± 1.9	0.41 ± 0.01	10.8 ± 0.04	10.6 ± 0.04
All plots	4.6 ± 0.03	4.4 ± 0.04	20.9 ± 0.5	25.2 ± 0.8	0.38/0.43 ± 0.01	10.9 ± 0.03	10.7 ± 0.04

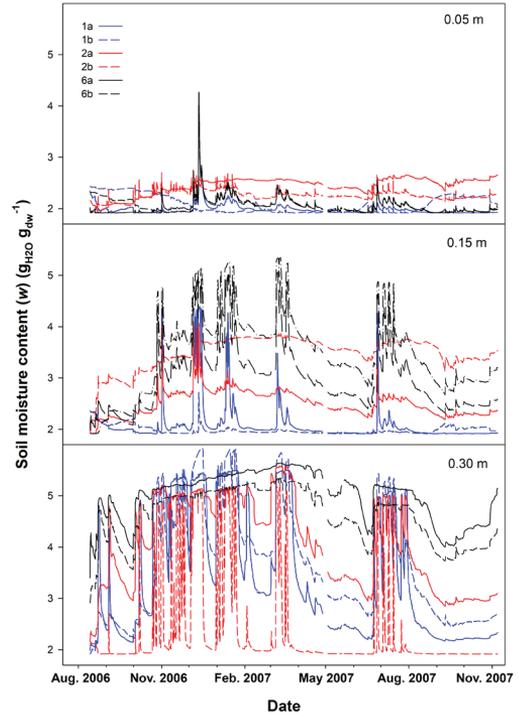


**Fig 6.** Mean soil moisture in six profiles at plots 1, 2 and 6, at the drained peat site in the Skogaryd research catchment from May to Aug. 2007.

ferred significantly (one-way ANOVA:  $F_5 = 80.8$ ,  $p < 0.001$ ) from all the other plots (Tukey HSD:  $p < 0.001$  for all; Table 2) and was 31 cm higher than the average for the other plots (Fig. 2). This was clearly due to its position in a depression on a forest machinery track, and we regarded plot 5 as an outlier with respect to groundwater table. The mean groundwater table was also significantly lower in plot 4 than at plots 2 and 6 (Tukey HSD:  $p = 0.013$  and  $p = 0.043$ , respectively), but the differences were relatively small (Table 2). When plot 5 was excluded from the analysis, the mean groundwater table at plot 4 was again significantly lower (one-way ANOVA:  $F_4 = 4.4$ ,  $p = 0.008$ ) than at plots 2 and 6 (Tukey HSD:  $p = 0.006$  and  $p = 0.022$ , respectively).

For soil moisture content, the vertical pattern in profile 2b deviated sharply from that of the other soil profiles where soil moisture was measured (Fig. 6). At 15 cm the soil was wetter in profile 2b than in profile 2a but at 30 cm, the soil was remarkably dry. At a depth of 15 cm, the soil in profile 2b also retained more of its moisture during dry periods compared with the other soil profiles (Fig. 7). At a depth of 5 cm the soil was wetter in both profiles in plot 2 than in the other plots (Fig. 6).

The C:N ratio of the soil did not differ significantly between plots for the 0–5 cm layer but for the 5–20 cm layer plot 5 had a significantly (one-way ANOVA:  $F_5 = 5.137$ ,  $p = 0.002$ ) higher C:N ratio than plot 4 (Games-Howell:  $p = 0.029$ ) (Table 2). The average C:N ratio for the two



**Fig 7.** Soil moisture at the depths of 5 cm, 15 cm and 30 cm, at the drained peat site in the Skogaryd research catchment, from 14 Aug. 2006 to 6 Nov. 2007.

sampled layers (0–5 cm and 5–20 cm), weighted for their different depths, was 24 (SE 0.62, range 18.5–34.5). The pH in the 0–5 cm layer was significantly (one-way ANOVA:  $F_5 = 6.2$ ,  $p < 0.001$ ) higher at plots 1 and 6, compared with plots 3 and 4 (Tukey HSD:  $p = 0.012$ ,  $p = 0.002$ ,  $p = 0.033$  and  $p = 0.005$  for 1 vs. 3, 1 vs. 4, 6 vs. 3 and 6 vs. 4, respectively). In the 5–20 cm layer, the pH was significantly higher (one-way ANOVA:  $F_5 = 13.7$ ,  $p < 0.001$ ) at plots 1, 5 and 6, compared with plots 2, 3 and 4 (Tukey HSD:  $p = 0.001$  for 1 vs. 2, 1 vs. 3, 1 vs. 4, 6 vs. 2, 6 vs. 3 and 6 vs. 4;  $p < 0.001$  for 5 vs. 2, 5 vs. 3 and 5 vs. 4) (Table 2). This indicated that the pH was higher in the southern than in the northern part of the site (Fig. 1).

## Discussion

### CO<sub>2</sub>

The CO<sub>2</sub> fluxes at the Skogaryd drained peat site often correlated well with soil temperature

at 5 cm depth. Similar results were obtained by Mäkiranta *et al.* (2008), examining afforested organic soil croplands in Finland, with soil temperature at a depth of 5 cm explaining 71–96 % of the temporal variation in soil respiration. Salm *et al.* (2012) found that 57–73% of the CO<sub>2</sub> effluxes of managed and natural peatlands was explained by water level and soil temperature at 10 cm. Although a large part of the soil respiration can be statistically explained by soil temperature, the autotrophic part of the soil respiration is, in reality, driven primarily by the supply of recent photosynthates to the roots (Högberg *et al.* 2001, Högberg and Read 2006) and is, therefore, more dependent on the seasonal pattern of carbon allocation to the roots than on temperature in itself (Högberg *et al.* 2001, Bhupinderpal-Singh *et al.* 2003). In the study by Mäkiranta *et al.* (2008), in which autotrophic and heterotrophic respiration were separated, these parts of the total respiration had identical seasonal patterns. We conclude that a large part of the variation in the CO<sub>2</sub> flux in our study was probably due to the variation in the supply of carbohydrates to the roots.

We compared the average annual CO<sub>2</sub> emission at the Skogaryd drained peat site, from autotrophic and heterotrophic soil respiration, with the values reported from other drained forested peatlands in the boreal and temperate regions (Appendix Table A1). The Skogaryd flux was higher than the fluxes measured at poorly drained forested Swedish peatlands by von Arnold *et al.* (2005a, 2005b, 2005c), as hypothesized. This is in agreement with the findings of Mäkiranta *et al.* (2009), based on measurements in Finland, that an optimum groundwater table for heterotrophic soil respiration originating from peat decomposition is 61 cm. In contrast, Widén (2001) presented CO<sub>2</sub> fluxes from a drained peatland in Sweden that were similar to those at the Skogaryd drained peat site, although that site was not well drained. Since heterotrophic and total soil respiration have been shown to increase with higher nutrient status (Silvola *et al.* 1996b, Minkkinen *et al.* 2007), we also expected our average annual CO<sub>2</sub> emissions at the Skogaryd drained peat site to be higher than those measured at drained forested peatlands with a lower nutrient status. However, the CO<sub>2</sub> fluxes at the

Skogaryd drained peat site were also higher than in previous studies on similar soils, i.e., forested organic soils with both a high nutrient status and a low groundwater table.

Higher CO<sub>2</sub> emissions from the soil at the Skogaryd drained peat site, compared with other studies, could have been due to greater biomass in the tree stand. The stand volume at the Skogaryd site was more than double that of the largest stand volumes in the studies described by Mäkiranta *et al.* (2007) and Minkkinen *et al.* (2007). Silvola *et al.* (1996a) and Maljanen *et al.* (2001) did not present values for stand volume, but the aboveground biomass of 40 600 kg ha<sup>-1</sup> mentioned in Maljanen *et al.* (2001) indicates that the tree stand volume was small. The higher flux values at the Skogaryd drained peat site could thus to some extent be the result of greater root respiration and a larger input of litter, compared with most other similar sites described in the literature. Silvola *et al.* (1996b) showed that the root contribution (including oxidation of root exudates and fine root litter) to soil respiration correlated well with tree stand volume in a study involving ten Finnish peatland sites. In addition, in a meta-analysis by Subke *et al.* (2006) using soil CO<sub>2</sub> flux partitioning studies carried out in many different ecosystems, it was shown that the relative contribution of autotrophic respiration to soil respiration increased with total soil respiration. The stand density, mean tree heights and diameters presented in Widén (2001) suggest a tree stand volume of less than 100 m<sup>3</sup> ha<sup>-1</sup>, i.e., much smaller than at the Skogaryd drained peat site. However, the C:N ratio (18) was lower than at the Skogaryd drained peat site, which may have led to higher heterotrophic fluxes (Silvola *et al.* 1996b, Minkkinen *et al.* 2007). In conclusion, it is likely that autotrophic respiration from the large tree stand contributed to the high CO<sub>2</sub> fluxes at the Skogaryd drained peat site.

Although a large tree stand could probably explain part of the discrepancy between our CO<sub>2</sub> flux results and those of previous studies, the measurement technique could also have played an important part. Heinemeyer *et al.* (2011) showed that the depth to which chambers are inserted into the soil has a drastic influence on measured CO<sub>2</sub> fluxes, even with insertion depths of only a few centimetres, due to severing of

fine roots. In the present study, great care was taken when chamber collars were installed, with the specific intention of leaving roots and mycorrhiza as undisturbed as possible. In the one previous study that yielded flux values close to those of the Skogaryd drained peat site (Widén, 2001), chambers were installed to a maximum depth of 0.5 cm, i.e., without any significant disturbance of roots or mycelia. In contrast, in the studies by von Arnold *et al.* (2005a, 2005b, 2005c), collars were installed several cm deep, which could have led to underestimations of the CO<sub>2</sub> flux. Mäkiranta *et al.* (2008) and Ojanen *et al.* (2010) installed their chambers in 2 cm deep grooves, in their non-trenched plots, which is not deep, but the installation may still have cut some mycorrhiza and fine roots and influenced CO<sub>2</sub> fluxes. Maljanen *et al.* (2001) did not specify chamber installation depth, while Silvola *et al.* (1996a, 1996b) and Laine *et al.* (1996) placed their chambers on top of the soil surface, using a cellular plastic seal. Ernfors *et al.* (2010), Klemedtsson *et al.* (2010) and Rütting *et al.* (2014) used the same method as in the present study, letting at least half of the chamber rest on top of the peat without any cutting. Our conclusion is that less cutting of roots and mycorrhiza may have added to the differences in CO<sub>2</sub> fluxes between the Skogaryd drained peat site and the sites described by von Arnold *et al.* (2005a, 2005b, 2005c), Mäkiranta *et al.* (2008) and Ojanen *et al.* (2010). However, it cannot explain the entire flux difference between the Skogaryd drained peat site and most other studies.

It is remarkable that the annual forest floor CO<sub>2</sub> fluxes measured at the Skogaryd drained peat site were about three times larger than that measured at the control plots of a wood ash fertilization study carried out about 200 m to the SW on the same drained peatland (Klemedtsson *et al.* 2010). This could not be explained either by site characteristics, which were similar, or by collar design or installation, which were the same at the two sites.

Similar differences were also found between the CO<sub>2</sub> fluxes measured at the Skogaryd drained peat site using automatic chambers and CO<sub>2</sub> emissions calculated from the manual samples collected at the same site for N<sub>2</sub>O and CH<sub>4</sub> flux calculations (data not shown). This discrepancy

was clearly due to properties of the chambers or the ways in which the chambers were used, since the fluxes were measured on the same collars. The main characteristics of the automatic chambers and their use at the Skogaryd drained peat site, that could differ from those of the manual chambers used and of those used at other sites, were: the use of fans and vents, the covering of the whole diurnal cycle, the closed chamber technique, the medium duration of the measurements and the fact that they were unusually high.

The use of fans in static chambers increases the gradient between soil and atmosphere during the measurement and can lead to overestimations of the gas flux when ambient wind speeds are low, e.g., during night-time (Lai *et al.* 2012, Koskinen *et al.* 2014). This is particularly likely to occur at porous soils such as peat and when using short measurement times (Lai *et al.* 2012). The automatic chambers used at the Skogaryd drained peat site were equipped with three fans, but the largest of the three was only in use before the measurement started, for the purpose of obtaining the same gas concentration in the chamber headspace as in the ambient air. This fan had a volumetric flow rate (32.1 L s<sup>-1</sup>) that was larger than the combined flow rate of the two fans used during the measurements (15.2 L s<sup>-1</sup>). Overestimation occurs when turbulence increases after closing the chamber (Lai *et al.* 2012, Koskinen *et al.* 2014) and in this case there would rather be a risk of underestimation of the flux, since the turbulence decreased when the measurement started. Also, CO<sub>2</sub> fluxes measured using the automatic chambers were higher during the day (06.00–18.00) than during the night (18.00–06.00), for the time period of 1 Aug. 2006 to 6 Nov. 2007 (paired t-test:  $t_{1047} = 9.644$ ;  $p < 0.001$ ). When analysing CO<sub>2</sub> fluxes for each month separately, night-time fluxes were significantly higher than daytime fluxes only in Jan. 2007. This makes it unlikely that the high fluxes were due to an overestimation caused by the fans. The soil at the Skogaryd drained peat site was also not particularly porous for a peatland, since it was drained and consisted of fen peat, and the height of the chambers allowed for relatively long measurement times, since headspace saturation was slower than it would have been for lower chambers. Compared

with manual measurements, which are for practical reasons almost exclusively carried out during daytime, measurements over complete diurnal cycles will give more realistic fluxes. Higher autotrophic CO<sub>2</sub> fluxes during the night-time have been observed in forest systems (Heinemeyer *et al.* 2011) and would have led to higher flux values for complete diurnal cycles than for daytime measurements. However, the opposite was true in our study and the measurement of full diurnal cycles could not explain why the fluxes were higher than in other studies.

Gas fluxes can be overestimated for closed dynamic chambers if there is a leakage in the pumping system or if the air flow back into the chamber is restricted relative to the air flow to the pump, causing a lower than ambient air pressure inside the chamber (Fang and Moncrieff, 1998). A well-designed vent should, however, effectively counteract this effect (Davidson *et al.* 2002). The vent used on the automatic chambers in this study conformed to the criteria by Hutchinson and Mosier (1981) and should have eliminated this type of overestimation. Some authors have debated whether a vent in itself could cause large overestimations of fluxes (Conen and Smith 1998, Hutchingson and Livingston 2001), due to a Venturi effect. This could have applied to our chambers, but a Venturi effect is larger when wind speeds are high, and there was no significant correlation in our study between hourly wind speed and CO<sub>2</sub> flux values, for the site as a whole. There was, however, a weak correlation between wind speed and CO<sub>2</sub> flux at plot 2, which means it is possible that the vents contributed to some extent to the high fluxes.

The chambers used in our study were considerably higher than those used in the other studies (see Appendix Table A1). The use of high chambers and short gas accumulation periods has been recommended in order to avoid underestimates of gas fluxes when using the closed chamber technique (Conen and Smith 2000, Healy *et al.* 1996). This could indicate that our high fluxes stemmed from a smaller underestimation than in most studies. The study by Widén (2001) yielded flux values comparable to those at the Skogaryd drained peat site, but they used an open chamber. Open chambers do not suffer from the effects of changes in flux gradi-

ents due to build-up of gas concentrations in the headspace (Pumpanen *et al.* 2004). Meyer *et al.* (2013) found an annual heterotrophic soil respiration of 47 600 (SD: 6200) kg CO<sub>2</sub> ha<sup>-1</sup> (including soil organic matter and litter respiration but excluding root respiration) in trenched plots at the Skogaryd drained peat site in 2008, using the same chambers as in the present study. They compared this value to a corresponding heterotrophic flux value of 33 000 kg CO<sub>2</sub> ha<sup>-1</sup> a<sup>-1</sup> calculated from micrometeorological measurements combined with biomass measurements and modelling. This independent confirmation of high heterotrophic forest floor CO<sub>2</sub> fluxes at the Skogaryd drained peat site strengthens the evidence that the high CO<sub>2</sub> fluxes are valid and that those measured by low manual unvented chambers at the same site (Ernfors *et al.* 2010, Klemetsson *et al.* 2010 & Rutting *et al.* 2014) were underestimated.

## CH<sub>4</sub>

Since soil water content is one of the most important factors controlling CH<sub>4</sub> uptake in soils (Ball *et al.* 1997, Jang *et al.* 2006), the low CH<sub>4</sub> uptake at plot 5 could be attributed to its high groundwater level. Plot 5 was regarded as an outlier with respect to groundwater table and was therefore regarded as an outlier also with respect to CH<sub>4</sub> fluxes. The relatively low groundwater table in plot 4 might explain the high CH<sub>4</sub> uptake at this plot, but in plots 1 and 6, the higher CH<sub>4</sub> uptake instead coincided with the higher pH in the southern part of the site. However, pH is only a minor factor controlling CH<sub>4</sub> uptake (Jang *et al.* 2006) and it is likely that the difference in pH mirrored differences in peat properties that were not analysed in this study, e.g., degree of decomposition, plant origin of the peat, peat structure or peat bulk density. These factors could affect e.g., the diffusivity of the soil, which would influence CH<sub>4</sub> uptake (King and Adamsen 1992, Dörr *et al.* 1993, Smith *et al.* 2000).

An uptake of CH<sub>4</sub> was expected, since the Skogaryd drained peat site is well drained and nutrient rich. Previous studies on organic forest soils with these characteristics have shown mean CH<sub>4</sub> uptake rates of, for example,

4.6 kg CH<sub>4</sub> ha<sup>-1</sup> yr<sup>-1</sup> (Maljanen *et al.* 2003), 1.3 kg CH<sub>4</sub> ha<sup>-1</sup> yr<sup>-1</sup> (Mäkiranta *et al.* 2007) and 0.8 kg CH<sub>4</sub> ha<sup>-1</sup> yr<sup>-1</sup> (Weslien *et al.* 2009). The CH<sub>4</sub> uptake rates of 5.9 kg ha<sup>-1</sup> a<sup>-1</sup> and 4.2 kg ha<sup>-1</sup> a<sup>-1</sup> predicted by the models by Minkkinen *et al.* (2007) and Ojanen *et al.* (2010), respectively, were close to the measured average site uptake, excluding plot 5, of 4.4 kg CH<sub>4</sub> ha<sup>-1</sup> (SE 0.41). Our measurements thus support the validity of these two models. The agreement between modelled and measured values also indicates that it would be possible to include data from the Skogaryd drained peat site in the Minkkinen *et al.* (2007) and Ojanen *et al.* (2010) models, which would broaden their ranges and increase their usefulness for up-scaling CH<sub>4</sub> fluxes to national levels.

## N<sub>2</sub>O

The sediment layer present at 25–30 cm depth in parts of plot 2 seemed to affect soil moisture content (particularly in soil profile 2b, Figs. 6 and 7) and in turn, the N<sub>2</sub>O emissions (Fig. 2). The high soil moisture content above the sediment layer and very low moisture below, and the fact that the soil moisture in profile 2b remained high during the relatively dry periods from Apr.–Jun. and from Aug.–Nov. 2007 (Fig. 7), led us to conclude that the sediment layer restricted vertical water movement and that the large N<sub>2</sub>O peak in Jul. 2007 was probably due to the combination of high soil temperature and high soil water content. The absence of a N<sub>2</sub>O peak in plot 2 in 2006, similar to that in 2007, could be explained by the different summer precipitation patterns. The summer of 2007 was extremely wet (SMHI 2008) and although most of the precipitation fell in July (Fig. 2), the soil moisture in the top soil layers of plot 2 remained high throughout the whole summer (Fig. 7). In 2006, the summer precipitation was lower than the long-term average (1961–1990) until the end of July (SMHI 2007). Due to this long period of drought, the soil never achieved particularly high moisture levels, despite heavy rains in August.

In contrast to plot 2, plot 5 had low N<sub>2</sub>O emissions, which was probably due to its position in a depression caused by forest machinery.

During parts of the year, the groundwater table in plot 5 would have been high enough to significantly reduce oxygen availability, which would have limited the aerobic process of nitrification. With lower nitrification, limited availability of nitrate as a substrate for denitrification could explain the lower N<sub>2</sub>O fluxes in this plot. It is also possible that a higher proportion of the total denitrification flux was in the form of N<sub>2</sub>, rather than N<sub>2</sub>O, due to the wetter conditions (Weier *et al.* 1993).

The N<sub>2</sub>O emission peak in Mar. 2007 is likely to have been induced by freezing and thawing, since air and soil temperature at 5 cm depth was around 0°C at this time (Fig. 5) and peaks in N<sub>2</sub>O emissions during freezing and thawing periods have often been observed (Goodroad and Keeney 1984, Christensen and Tiedje 1990, Flessa *et al.* 1995; Röver *et al.* 1998, Dörsch and Bakken 2004). The mechanism behind these emissions is not yet clear (Öqvist *et al.* 2004), but contributing factors could include the release of physically trapped N<sub>2</sub>O (Goodroad and Keeney 1984) and the increased supply of readily available organic carbon to the microbial population, which also enhances anaerobiosis (Mørkved *et al.* 2006).

The average annual N<sub>2</sub>O fluxes from the Skogaryd drained peat site were similar to those measured by Klemmedtsson *et al.* (2010), in the control plots of an ash treatment experiment on the same peatland (5.0 kg N<sub>2</sub>O ha<sup>-1</sup> and 3.2 kg N<sub>2</sub>O ha<sup>-1</sup>, in 2006–2007 and 2007–2008, respectively). The average N<sub>2</sub>O flux presented in Ernfors *et al.* (2011), which was also used in the calculations by Meyer *et al.* (2013), was higher than in the present study, since it was based on only three of the plots, including the high-emitting plot 2. The average N<sub>2</sub>O emission for the site, excluding plot 2, (2.7 kg N<sub>2</sub>O ha<sup>-1</sup> a<sup>-1</sup>, SE 0.23) was in between the values predicted on the basis of the C:N ratios of the 0–5 cm and 0–2 cm soil depths, using the model presented by Ernfors *et al.* (2007), and the C:N ratios used to build the model referred to the upper 10–20 cm of the soil. Ojanen *et al.* (2010) concluded that the C:N ratio was the factor that best explained N<sub>2</sub>O emissions, but all the three models that they constructed predicted much lower emissions (0.8–1.7 kg N<sub>2</sub>O ha<sup>-1</sup> a<sup>-1</sup> for

the 0–5 cm layer and 0.3–1.2 kg N<sub>2</sub>O ha<sup>-1</sup> a<sup>-1</sup> for the 0–20 cm layer) for the Skogaryd drained peat site, compared with both measured fluxes and the Ernfors *et al.* (2007) model. Using the Skogaryd drained peat site C:N ratios with the model constructed by Klemmedtsson *et al.* (2005), which was based on exactly the same data as the Ernfors *et al.* (2007) model, the predicted flux values were also lower than the measured values (0.4–1.4 kg N<sub>2</sub>O ha<sup>-1</sup> a<sup>-1</sup>). In contrast to the other models mentioned here, which all take the form of:  $y = y_0 + ae^{-bx}$ , the Ernfors *et al.* (2007) model is written in the form of:  $y = e^{(a - b(\ln x))}$ , which created a slightly flatter curve when the model was fitted to the data. Our study confirms that soil C:N ratios has the potential to be a good predictor of N<sub>2</sub>O emissions from drained forested organic soils, but that the choice of model can have a large influence on the results. Including more data as it becomes available, especially annual N<sub>2</sub>O flux values from sites with low C:N ratios, is likely to substantially improve these types of models.

The N<sub>2</sub>O emission peak in plot 2 in 2007 showed that there is a potential for high summer N<sub>2</sub>O emissions at the Skogaryd drained peat site, if the soil moisture content is sufficiently high. This suggests that the annual N<sub>2</sub>O emissions are limited by the low groundwater table. Jungkunst *et al.* (2008) found in a microcosm experiment that the maximum N<sub>2</sub>O flux from hydromorphic soils (including one Histosol) occurred when the groundwater table was at a depth of 20 cm. Goldberg and Gebauer (2009) also found that drought turned a Haplic Podsol with a 6–10 cm mor layer into a sink for N<sub>2</sub>O, while rewetting of the soil turned it back into a source. These studies support the conclusion that the summer N<sub>2</sub>O emissions at the Skogaryd drained peat site were limited by drought. The N<sub>2</sub>O fluxes from the well-drained sites studied by Mäkiranta *et al.* (2007) and Weslien *et al.* (2009) were higher than those at the Skogaryd drained peat site, despite lower or similar mean groundwater tables (~40–50 cm and 53 cm, respectively). However, these sites also had low C:N ratios (13–28 and 13, respectively), which suggests that the soil C:N ratio is the main factor determining N<sub>2</sub>O emissions and that the groundwater table is a secondary, regulating, factor. The use

of the model presented in Ernfors *et al.* (2007) is supported by the measured data, but the inclusion of groundwater levels in the model would likely improve its predictions.

## Representativeness of the site

The Skogaryd drained peat site can be assumed to be representative of Swedish peatlands that were intensively cultivated prior to afforestation. The site was used for growing oats and potatoes for approximately 80 years and the cultivation was therefore both intensive and long-term. The current area of afforested agricultural soils in Sweden cannot be accurately retrieved from any database, but indirect estimates indicate that these soils cover large areas. In the 1920s, approximately 600 000 ha of organic soil was used for agriculture in Sweden (Hallgren and Berglund 1962) while presently this area is only 301 489 ha (Berglund and Berglund 2010). These estimates include both peat and "gyttja" soils, but peat soils dominate (Berglund and Berglund 2010). The approximately 300 000 ha of organic soil that appear to have been taken out of agricultural use can largely be assumed to have been afforested. In the Swedish National Forest Soil Inventory (RIS 2019b), there are sample plots where the soil can be classified as organic (C content > 12%) but not as peat, since the top soil is mixed with mineral soil and classified as an A horizon. The area of this category of organic soil was calculated by Ernfors *et al.* (2007) and found to cover 169 000 ha in Sweden, amounting to 11% of the total drained organic forest soil. Ernfors *et al.* (2007) assumed that most of this area was former agricultural soil, since the mixing of mineral soil and organic matter can be the result of tillage. Soils which had a thin peat layer before drainage and tillage, or soils which have lost most of their peat due to decomposition and then been tilled, would probably belong to this category. Soils with a thicker peat layer are less likely to have been mixed with mineral soil, and large areas of afforested former agricultural soil could therefore also have been categorised as true peat soils. Most likely, former agricultural soils constitute a significant part of the 1.5 Mha

(Ernfors *et al.* 2007) of drained organic forest soils in Sweden.

According to the Swedish National Inventory of Forests (RIS 2019b) the median top soil C:N ratio for all drained organic forest soils in Sweden is 24. This value is based on samples from the 0–10 cm layer if the top soil is an A horizon and the 0–30 cm layer if the top soil is an O horizon (RIS 2019a). The Skogaryd drained peat site with its mean C:N ratio of 24 in the upper 20 cm can thus be regarded as representative of Swedish drained organic forest soils in this respect. The median C:N value for the organic soils not classified as peat in the Swedish National Forest Soil Inventory is 21, based on samples from the 0–10 cm layer. This means that the Skogaryd drained peat site, with a C:N ratio of 21 in the 0–5 cm layer, can be seen as representative also for this type of soils, which are likely to have been cultivated prior to afforestation (Ernfors *et al.* 2007).

In the Swedish National Inventory of Forests, the Swedish forest soils are divided into five soil moisture classes: dry, mesic, mesic-moist, moist or wet (RIS 2019a). According to the calculations by von Arnold *et al.* (2005a), which were based on the Swedish National Forest Inventory (part of the Swedish National Inventory of Forests, RIS 2019b), 80% of the Swedish drained organic forestland belongs to the soil moisture class mesic-moist. The mesic-moist soil moisture category is assumed to represent sites with an average growing season groundwater table between 0.5 m and 1 m, based on observations of e.g., topography, visible surface water and vegetation (RIS 2019a). The average groundwater level for the Skogaryd drained peat site, between 1 Aug. 2006 and 31 Jul. 2007, was 43 cm (SE 0.01) if plot 5 was excluded (Table 2). During the 2007 growing season, the average groundwater table (excluding plot 5) was at a depth of 49 cm. Since the summer of 2007 was extremely wet in southern Sweden, we assumed that the long-term average groundwater table was lower than in 2007 and the site should therefore be classified as mesic-moist. This was also consistent with observations of visible surface water and vegetation. Being classified as mesic-moist, the site is representative of most of the Swedish drained organic forest land, with

respect to moisture status. Soils in this soil moisture class can be considered as well drained (von Arnold *et al.* 2005a).

In summary, the Skogaryd drained peat site is representative of well-drained organic forest soils in Sweden and has a background which places it in the category of afforested agricultural peatlands. It would therefore be well justified to use results from the site in predictions of national N<sub>2</sub>O emissions from well-drained afforested agricultural peatlands.

## Conclusions

The total annual CH<sub>4</sub> flux for an afforested peatland in the Skogaryd research catchment was in agreement with the models presented by Minkkinen *et al.* (2007) and Ojanen *et al.* (2010), which predict annual CH<sub>4</sub> fluxes of drained forested peatlands on the basis of their stand biomass. The total annual N<sub>2</sub>O flux was also in agreement with the flux predicted from the C:N ratio of the soil using the model presented in Ernfors *et al.* (2007). The prediction power of such models was thus supported. The spatial and temporal patterns of N<sub>2</sub>O emissions suggested that these were limited by drought during the summer. The total annual soil CO<sub>2</sub> emissions were large in comparison to measurements made at Swedish peatlands with higher groundwater tables (von Arnold *et al.* 2005a, 2005b, 2005c), as hypothesized, but also in comparison to measurements at well-drained forested peatlands with a high nutrient status (Silvola *et al.* 1996a, Maljanen *et al.* 2001, Mäkiranta *et al.* 2007, Mäkiranta *et al.* 2008). The high emissions could partly be attributed to root respiration and decomposition of recent litter from the productive tree stand, but we concluded that the results were also affected by an improved chamber design. This suggests that further comparisons and standardizations of static chambers are needed.

*Acknowledgements:* We are sincerely grateful to the late Mats Rantfors for his technical assistance during the project. We acknowledge the financial support from FORMAS, the Swedish Research Council, Tellus (The Centre of Earth Systems Science, University of Gothenburg) and the European project, NitroEurope.

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## Appendix

**Table A1.** Annual CO<sub>2</sub> emissions reported from drained forested peatlands in boreal and hemiboreal areas, measured using open or closed chambers. Plots fertilized for experimental purposes are not included.

No	Forest type	Location	Chamber type	Chamber installation depth (m)	Chamber height (m)	C/N or general nutrient status	Groundwater water table (m)	CO <sub>2</sub> flux (kg CO <sub>2</sub> ha <sup>-1</sup> a <sup>-1</sup> )	Method for winter fluxes	Flux type	Authors
1	Pine-dominated	Southern Finland	Open chambers	On top of soil	0.05	22–90	0.24–0.33 (growing season)	8800–14 300	Winter fluxes calculated using March 1992 values	Soil respiration (forest floor vegetation removed)	Laine <i>et al.</i> 1996
2	Pine, birch and not specified	Southern Finland	Open chambers in summer, closed in winter	On top of soil	0.05 in summer, 0.2 in winter	Ombrotrophic to mesotrophic	0.14–0.55	5100–22 200	Winter fluxes from another, similar, site used for some sites	Soil respiration (forest floor vegetation removed)	Silvola <i>et al.</i> 1996a
3	Pine- and spruce-dominated	Central Sweden	Open chambers	0.005	0.3 at highest point	High and medium productivity, C/N 18 for the medium site	0.7 and 0.17	27 900 and 65 000	Winter flux partly modelled using soil temperature	Forest floor flux, transparent chamber	Widén 2001, Morén and Lindroth 2000, von Arnold <i>et al.</i> 2005a, M. Lundblad (pers. comm.)
4	Spruce-dominated	Southern Sweden	Closed chambers	Several cm	ca. 0.1–0.2	High productivity	0.12–0.60	11 000–19 000	Whole year measured	Forest floor respiration	von Arnold <i>et al.</i> 2005a
5	Birch and alder	Southern Sweden	Closed chambers	0.04–0.1	0.01–0.4	16–22	0.15–0.18	15 000–22 000	Whole year measured	Forest floor respiration	von Arnold <i>et al.</i> 2005b
6	Spruce- and pine-dominated	Southern Sweden	Closed chambers	Several cm	ca. 0.1–0.2	26–40	0.17–0.27	9 400–19 000	Whole year measured	Forest floor respiration	von Arnold <i>et al.</i> 2005a, 2005c
7	Birch-dominated	Southern Sweden	Closed chambers	Several cm	ca. 0.1–0.2	13	0.48–0.7	22 000–28 000	Whole year measured	Forest floor respiration	Weslien <i>et al.</i> 2009, von Arnold <i>et al.</i> 2005a

Table A1. (continued)

No	Forest type	Location	Chamber type	Chamber installation depth (m)	Chamber height (m)	C/N or general nutrient status	Groundwater water table (m)	CO <sub>2</sub> flux (kg CO <sub>2</sub> ha <sup>-1</sup> a <sup>-1</sup> )	Method for winter fluxes	Flux type	Authors
8	Pine-dominated	Estonia and Finland	Closed chambers	0.3 (trenching)	0.15 or 0.17, on top of snow in winter	Ombrotrophic to eutrophic	ca. 0.14–0.9	9100–18 900	Whole year measured	Heterotrophic respiration (trenched plots, litter and forest floor vegetation removed)	Minkkinen <i>et al.</i> 2007
9	Birch, pine alder	Southern and central Finland	Closed chambers	0.3 (trenching)	0.15 on top of snow in winter	13–28	ca. 0.3–0.8	7600–19 800	Whole year measured	Heterotrophic respiration (trenched plots, litter and vegetation removed) Afforested agricultural and cutaway sites	Mäkiranta <i>et al.</i> 2007, 2008
10	Pine-dominated	Southern Sweden	Closed chambers	0–0.05	0.06–0.19	33–36	0.15–0.57 (growing season)	6700–14 800	Whole year measured	Forest floor respiration	Ernfors <i>et al.</i> 2010
11	Pine- and birch-dominated	Estonia	Closed chambers	Not specified	0.5	Ombrotrophic to eutrophic	0–0.15	1600–12 500	Whole year measured	Forest floor respiration	Salm <i>et al.</i> 2012
12	Pine-dominated	Northern Finland	Closed chambers	0.02, but trenched to 0.3	0.15, on top of snow in winter	Meso-trophic	Not specified	8700	Whole year measured	Heterotrophic respiration (trenched plots, litter and forest floor vegetation removed)	Moilanen <i>et al.</i> 2012

Table A1. (continued)

No	Forest type	Location	Chamber type	Chamber installation depth (m)	Chamber height (m)	C/N or general nutrient status	Groundwater water table (m)	CO <sub>2</sub> flux (kg CO <sub>2</sub> ha <sup>-1</sup> a <sup>-1</sup> )	Method for winter fluxes	Flux type	Authors
13	Spruce-dominated	Southern Finland	Closed chambers	0–0.05 but trenched to 0.6	ca 0.92	24	0.45	47 600	Whole year measured	Heterotrophic respiration (trenched plots, forest floor vegetation removed)	Meyer <i>et al.</i> 2013
14	Not specified (68 sites used)	Finland	Closed chambers	0.02	0.15	16–45	0.01–0.8 (May to Oct.)	11 650–44 400	Winter fluxes modelled based on growing season fluxes and soil temperatures	Afforested agricultural site Forest floor respiration	Ojanen <i>et al.</i> 2010, 2013
15	Spruce-dominated	Southern Sweden	Closed chambers	< 0.02	ca 0.10	23	0.3–0.8	17 000–17 600	Whole year measured	Forest floor respiration	Klemetsson <i>et al.</i> 2010
16	Spruce-dominated	Southern Sweden	Closed chambers	< 0.02	ca 0.10	23	0.25–0.7	13 700–15 300	Snow-season flux estimated by interpolation	Forest floor respiration	Rütting <i>et al.</i> 2014
17	Pine and birch	Southern Finland	Closed chambers	0.02–0.03	0.15–0.25	34–38	0.42 in summer, ca. 0.05 in winter	20 600–23 100	Whole year measured	Forest floor respiration	Minkkinen <i>et al.</i> 2018, Ojanen <i>et al.</i> 2012
18	Pine, birch and spruce	Central and northern Finland	Closed chambers	0.02, but trenched to 0.5	0.15	Ombrotrophic to eutrophic	ca. 0.12–0.58	ca. 5500–15 500	Snow-season flux modelled using soil temperature	Heterotrophic respiration (trenched plots, forest floor vegetation removed)	Ojanen <i>et al.</i> 2019