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Soil CO_2 CH_4 and N_2O fluxes from an afforested lowland raised peatbog in Scotland: implications for drainage and restoration

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Abstract. The effect of tree (lodgepole pine) planting with and without intensive drainage on soil greenhouse gas (GHG) fluxes was assessed after 45 yr at a raised peatbog in West Flanders Moss, central Scotland. Fluxes of CO_2 CH_4 and N_2O from the soil were monitored over a 2-yr period every 2 to 4 weeks using the static opaque chamber method in a randomised experimental block trial with the following treatments: drained and planted (DP), undrained and planted (uDP), undrained and unplanted (uDuP) and for reference also from an adjoining near-pristine area of bog at East Flanders Moss (n-pris). There was a strong seasonal pattern in both CO_2 and CH_4 effluxes which were significantly higher in late spring and summer months because of warmer temperatures. Effluxes of N_2O were low and no significant differences were observed between the treatments. Annual CH_4 emissions increased with the proximity of the water table to the soil surface across treatments in the order: DP < uDP < uDuP < n-pris with mean annual effluxes over the 2-yr monitoring period of 0.15, 0.64, 7.70 and 22.63 g CH_4 m^{-2} yr^{-1}, respectively. For CO_2, effluxes increased in the order uDP < DP < n-pris < uDuP, with mean annual effluxes of 1.23, 1.66, 1.82 and 2.55 kg CO_2 m^{-2} yr^{-1}, respectively. CO_2 effluxes dominated the total net GHG emission, calculated using the global warming potential (GWP) of the three GHGs for each treatment (76–98 %), and only in the n-pris site was CH_4 a substantial contribution (23 %). Based on soil effluxes only, the near pristine (n-pris) peatbog had 43 % higher total net GHG emission compared with the DP treatment because of high CH_4 effluxes and the DP treatment had 33 % higher total net emission compared with the uDP because drainage increased CO_2 effluxes. Restoration is likely to increase CH_4 emissions, but reduce CO_2 effluxes. Our study suggests that if estimates of CO_2 uptake by vegetation from similar peatbog sites were included, the total net GHG emission of restored peatbog would still be higher than that of the peatbog with trees.

1 Introduction

Globally, undisturbed peatlands are important sinks for atmospheric carbon dioxide (CO_2) (Alm et al., 1997; Turunen et al., 2002), but emit methane (CH_4) and the net global warming impact may be near zero (Cannell et al., 1993). Microbial production of CH_4 is strictly anaerobic, production of CO_2 aerobic and N_2O can be produced under both aerobic and anaerobic conditions, and it may be consumed in wet, nitrogen-poor soils (e.g., Chapuis-Lardy et al., 2007). Therefore, the production and consumption of these greenhouse gases (GHG) in peat soils is highly dependent on the oxygen availability in the soil and, thus, the depth of the water table (Martikainen et al., 1993; Aerts and Ludwig, 1997). The importance of managed peatlands in the global carbon budget and in the GHG radiative forcing of climate is uncertain because of the contrasting effects of water table/aerobicity conditions and temperature on CO_2 and CH_4 fluxes (Oechel et al., 1993; Laine et al., 1996; Shindell et al., 2004; Ise et al., 2008) and the supply of readily decomposed substrate (Christensen et al., 2003; Sirin and Laine, 2008). Particular peatland vegetation components could also provide a direct route for methane release to the atmosphere by bypassing the oxidation layer and methanotrophs, thus, increasing emission rates (e.g., Nilsson et al., 2001; Sirin and Laine, 2008;

Couwenberg, 2009). As a result, vegetation and microtopography are strong predictors of emission rates (Bubier et al., 1995), and CH_4 fluxes can vary more within a few metres than across peatland regions (Moore et al., 1998). In peat soils, CH_4 is only produced when labile carbon substrates are amply available (Couwenberg, 2009) and old (recalcitrant) peat components play only a minor role as a substrate for CH_4 production (e.g., Charman et al., 1999; Clymo and Bryant, 2008). Although many northern peatlands are a suitable habitat for anaerobic CH_4-producing bacteria, net CH_4 fluxes are typically low in forested systems (Coles and Yavitt, 2002).

It is estimated that the UK peatland area of 2.3 Mha contains about 2.2 billion t of carbon, 68 % of which is in the top 0–100 cm soil layer and the reminder is in deep peats > 100 cm deep (Billett et al., 2010). Drainage for forestry affects the hydrology of peatlands (e.g., King et al., 1986; Hillman, 1992) and, thus, could have a strong impact on the production and consumption processes and fluxes of GHGs in afforested peatland. Peatland drainage virtually stops methane emission and increases CO_2 loss through aerobic decomposition, but can also increase carbon fixation by the peatland vegetation partly because of the stimulation caused by microbial mineralisation of nitrogen, resulting in either a net loss or gain in carbon (Cannell et al., 1993). In northern latitudes, higher CO_2 emissions (von Arnold et al., 2005a, b, c) and one to several orders of magnitude lower CH_4 emissions (Laine et al., 1996) were observed from drained fen and bog peatland sites. According to Minkkinen and Laine (1998) enhanced tree stand growth in some cases after drainage can compensate for the carbon loss from peat. Comparison of the average annual CO_2 emissions in drained and undrained afforested blanket peat in Ireland revealed no clear pattern in relation to drainage (Byrne and Farrell, 2005) and suggested that afforestation does not always lead to an increase in soil CO_2 emissions. Those authors also concluded that losses of soil C are compensated by C uptake by the trees. Hargreaves et al. (2003) measured the net CO_2 exchange over undisturbed and drained afforested sites of different ages and suggested from modelled C balances that afforested peatlands in Scotland accumulate more carbon in trees, litter, soil and forest products than is lost from the peat between 90 and 190 yr, depending on the rate of peat loss.

Concern has been expressed (e.g., Thompson, 2008) that restoration of peatlands is promoted as a means of restarting their carbon sink function, but that, until recently, CH_4 emissions have not been considered when estimating restoration benefits (Baird et al., 2009). A rise in the water depth (e.g., from seasonal variation, after clearfelling or after drain blocking for peatland restoration) can increase CH_4 emission (e.g., Funk et al., 1994; Aerts and Ludwig, 1997), but peat temperature may also increase, particularly in colder climates (e.g., Prévost et al., 1999; Huttunen et al., 2003) and, thus, it may cause higher CO_2 emissions. In contrast, van den Bos (2003) indicated that wetland restoration of reclaimed peat areas in the western Netherlands led to a reduction of GHG emissions because the expected increase in anaerobic production of CH_4 is much smaller than the decrease in aerobically produced CO_2. Also, although drainage decreases CH_4 efflux, rewetting does not necessarily lead to an immediate rise in CH_4 emission (Tuittila et al., 2000).

Although peatland conservation and restoration is a high priority under current biodiversity protection objectives, its impact on total GHG and soil carbon budgets requires further quantification. The recent comprehensive review by Worrall et al. (2011) concluded that many restoration or management interventions may not provide a benefit in terms of GHG emissions because the flux of CH_4 is often a more important component of the C balance of restored peatlands when considered in terms of global warming potential than the net exchange of CO_2. According to the recent report by the UK Joint Nature Conservation Committee (Birkin et al., 2011) there is a need to produce robust, accurately-quantified GHG emission factors for peatlands under both existing steady management states and during transitions, with field research required to improve comparisons and fill evidence gaps. The aim of this study was to monitor soil CO_2, CH_4 and N_2O fluxes from a raised peatbog to (i) quantify the long-term effects of afforestation with and without intensive drainage; (ii) compare the soil GHG fluxes with those of a near-pristine peatbog area nearby (to address possible consequences of restoration), and to (iii) determine the influence of environmental variables (temperature, water table depth and water chemistry) on the GHG fluxes.

2 Site description and experimental layout

The overall experimental area, about 400 ha, was located in Flanders Moss Forest (15 m above sea level; 56°08′ N, 4°18′ W; British National Grid reference NS 568 959) which occupies West Flanders Moss (WFM), one of a group of lowland ombrotrophic raised bogs covering some 1620 ha and formed on the uplifted former estuary of the River Forth in the Carse of Stirling in Central Scotland. The Moss, which had been drained by ditches dug by hand to 0.6 m depth at 8–10 m intervals during the 1920s to improve its condition for grouse shooting, was ploughed and afforested by the Forestry Commission in 1965. It was planted with lodgepole pine (*Pinus contorta* Dougl. var. *latifolia* Engelm.) to give 8900 trees per ha. The 2001 forest district inventory for the experimental area showed mean tree diameter at breast height (dbh) of 17 cm, Yield Class of 10 m^3 ha^{-1} yr^{-1} and mean top height of the trees of 18.1 m; some of the trees had fallen due to wind-throw. The soil is organic-rich with lightly humified, fibrous *Sphagnum/Eriophorum* peat up to 8.5 m deep (average 4.6 m) over estuarine clay. The pre-planting peat analysis of the 15–45 cm layer showed Ash 1.7 % oven dry wt, N 1.4 %, P 0.021 %, and K 0.009 % (J. B. Craig,

Macaulay Institute for Soil Research, personal communication, 1964).

The work reported here was carried out in a forestry drainage experiment covering 46 ha dating from the original planting in 1965 laid out by the then Forestry Commission Research Directorate. Eight treatments involving different types, intensity and depth of drainage plus an undrained, unplanted control had been laid out in 0.5 ha plots in four randomised blocks to investigate their effects on tree growth, stability, rainfall interception and soil aeration (Lees, 1972). For the purpose of the current experiment, three treatments within each of the four randomised blocks were selected: cross drained at 7.6 m spacing to 1.2 m depth and planted (DP); undrained and planted (uDP); and undrained and unplanted although this may be affected to some degree by the drying effects of the surrounding forest (uDuP). A 20×30 m plot on a separate bog, East Flanders Moss (EFM, Flanders Moss National Nature Reserve, grid reference NS 646 979, 7.5 km to the east) was also used to provide a "near-pristine bog" reference (n-pris). This 20×30 m plot was within the 40 ha former Polder Plantation which was afforested in 1962, felled in 1998 and subsequently restored to active raised bog by blocking the drains. For unknown reasons this plot had never been ploughed or planted during these previous land cover changes. Although it will have been affected by the surrounding forest (e.g., increased shelter, decreased light and lowered water table), it had retained a good cover of *Sphagnum* mosses and other bog vegetation and had become extremely wet when the surrounding plantation was felled and the wider bog area restored.

3 Methods

3.1 Gas flux measurements and analysis

Surface CO_2, CH_4 and N_2O fluxes were measured using the manual static chamber method, with opaque PVC chambers ($0.4 \times 0.4 \times 0.25$ cm) placed on permanently installed collars. A total of 40 collars was inserted tightly to a depth of 3 cm into the ground prior to the start of measurements; three replicate collars per treatment per block at the WFM site and four replicate collars at the EFM site about 3 m apart. Generally, collars were positioned randomly, but in the afforested plots the collars, where possible, were positioned to sample the range of soil surface variations caused by ploughing prior to planting (i.e., ridge, furrow and original surface). The top of each collar (which was kept level) had a water channel to ensure a gas-tight seal between the collar and chamber. During each gas flux measurement, chambers were placed on top of the collars for 60 min and duplicate gas samples of the chamber headspace were taken at 3 or 4 times (0, 30 and 60 min or 0, 20, 40, 60 min at the EFM site) after chamber closure by connecting a polypropylene syringe to the chamber sampling port fitted with a three-way stopcock. The syringes were immediately used to fill (under atmospheric pressure) pre-evacuated 20 mL vials fitted with Chlorobutyl rubber septa.

Concentrations of CO_2, CH_4 and N_2O were determined using a headspace-sampler (TurboMatrix 110) and gas chromatograph (Clarus 500, PerkinElmer) fitted with two identical $30\,m \times 30$ mm internal diameter megabore capillary porous Layer Open Tubular columns (Elite PLOT Q) maintained at 35 °C. The chromatograph was equipped with an electron capture detector (ECD) operated at 350 °C for N_2O analysis, a flame ionisation detector (FID) operated at 350 °C for CH_4 analysis and a catalytic reactor (methanizer) to reduce any CO_2 in the sample to CH_4 before analysis by the FID detector. Peak areas were estimated using a PerkinElmer integrator and results were calculated from detector responses to calibration mixture standards of 0.2–5 ppm N_2O, 1.2–30 ppm CH_4, and 300–7500 ppm CO_2. Fluxes were calculated from the linear increase of gas concentrations inside the chamber with time. The linearity was confirmed at the start of the experiment by measuring concentrations of the gases at 0, 5, 10, 20, 40 and 60 min after chamber closure.

In this method, the CO_2 flux was that from aerobic and anaerobic decomposition processes, respiration of other soil organisms, total dark respiration of ground vegetation and root respiration of trees. Because of the long distance between the different treatments, the gas sampling took place over two days, one to sample from all the randomised experimental treatments at WFM (generally between 09:00–17:00 h in a systematic order) and one for the gas sampling at EFM (between 10:00–12:00 h). Therefore, some effects on the results may be expected due to diurnal variations within the experimental treatments at WFM (i.e., between DP, uDP and uDuP treatments) and day-to-day climatic variations between those and EFM site. Flux measurements were conducted every two weeks in the first year between February 2008 and February 2009. In the second year, fluxes were measured monthly up to December 2009 after which the measurements were stopped because a heavy snowfall made it impossible to locate the chamber frames.

3.2 Environmental monitoring

Water table depth (cm from ground surface) was measured from dipwells (one per treatment in each block) inserted to a depth of 100 cm. Dipwells consisted of 6 cm diameter high-density polyethylene pipes with slots along the pipe length and screw caps (Merton Geotechnical Services Ltd., Bury St Edmunds, Suffolk, UK) to prevent rain entering. During each sampling day, water depths were measured across the sites using a water dip-meter (DIP 30, Geosense, Merton Geotechnical Services Ltd.) and soil temperatures at 1, 5 and 15 cm depth were measured manually with a digital temperature probe. The soil temperature at 1, 5 and 15 cm was also measured continuously from two plots (uDuP and uDP) throughout the experimental period using temperature

probes connected to a data logger (21X Micrologger, Campbell Scientific Ltd., Shepshed, Leics, UK). However, due to data logger failure there were gaps in the results. Therefore, daily climatic data from a nearby area was obtained from the British Atmospheric Data Centre (BADC) for precipitation (Auchentroig Estate, about 3.5 km away from the site; grid reference NS 544 934, elevation 46 m) and for air temperature (Portnellan Farm, Gartocharn, about 18 km from the site; grid reference NS 402 868, elevation 40 m). Water samples were taken from each dipwell during the gas flux measurements and analysed for dissolved organic nitrogen (DON) and dissolved organic carbon (DOC) by combustion method using Thermalox analyzer (Analytical Sciences UK, Cambridge, UK) and pH by probe (InLab science Pro, Mettler Toledo Ltd, Leicester, UK). At the end of the experiment, samples of the peat were taken at 0–10 cm and 10–20 cm depth from an area close to each chamber using a 5×5 cm corer, and were analysed for total C, total N, pH and bulk density. The pH was measured in a 1 : 5 soil-to-water suspension by a pH probe (Thermo Electron Corporation, USA), bulk density (g cm^{-3}) was determined by dividing the weight of oven-dried samples by their volume and the total C and N were determined by a combustion method in an elemental analyser (Carlo Erba Flash EA1112, CE Instruments Ltd, Wigan, UK).

3.3 Statistical analysis

Experimental treatments at WFM were set out in a randomised block design of 4 blocks $\times$ 3 treatments and within each treatment there were 3 replicated flux chambers (i.e., a total of 36 chambers at WFM). These were compared with a single block of 4 replicated chambers at EFM. Fluxes within replicated chambers were, on some occasions, skewed by high individual values that were considered to be accurate with no evidence of nonlinearity or ebullition and, therefore, all analyses were based on the median values of the 3 or 4 replicates. Annual cumulative fluxes of CO_2, CH_4 and N_2O from each treatment and block were quantified by calculating the mean of the measurements on two succeeding sample dates and multiplying it by the number of elapsed days between the dates over the total monitoring period in year 1 (2008) and year 2 (2009). The annual water table depth was estimated using the mean of all measurements per year. Analysis of variance (ANOVA) was then used to determine significant differences in fluxes and water depth between treatments within each year and over the study period.

To identify the most significant factors driving gas emissions, linear mixed models were fitted to the 2-weekly and 4-weekly flux measurement plot data in year 1 and 2, respectively, and linked to recorded environmental factors, i.e., plot temperature, rainfall, water depth and water chemistry (DOC, DON and pH). As part of the modelling process, environmental variables and their interactions were treated as fixed-effects whilst the repeated measure and randomised block design of the experimental design required the fitting of random-effects to account for likely correlations between observations within the same plot and observations taken on the same assessment date. For each gas, a series of linear mixed models were fitted and subsequently simplified by removing non-significant variables, factors and interaction terms. In addition, model fitting was improved by applying log and square root transformations to observed CH_4 and CO_2 fluxes, respectively, (the occasional negative flux for CH_4 was resolved by adding a constant of 2.507 to all values) and removing four extreme outliers (< 1 % of total data) from the methane dataset. All statistical analyses were undertaken using either Genstat (Payne, 2009) or SAS (SAS Institute Inc, 2008) statistical software.

As part of the linear mixed modelling process described above, random-effect parameters are estimated with an average effect size of zero. Consequently, the application of results to other similar peatland areas can be achieved by using observed site variables and fixed-effect parameter values estimated from the CO_2 and CH_4 models.

For CO_2 the model equation simplifies to:

$$F_{\text{est}} = T_{15} \cdot y + 1.1, \quad (1)$$

where F_{est} is the square root of the estimated CO_2 efflux (g m^{-2} d^{-1}); T_{15} is the observed soil temperature at 15 cm depth; y are the model parameters for the treatment-specific temperature coefficients (0.100 DP, 0.076 uDP, 0.160 uDuP, 0.117 n-pris).

For CH_4:

$$F_{\text{est}} = T_{15} \cdot y + \text{wtd} \cdot z + i, \quad (2)$$

where F_{est} is the natural log of the estimated CH_4 flux (mg m^{-2} d^{-1}) + 2.507; y are the model parameters for the treatment-specific temperature coefficients (0.010 DP, 0.021 uDP, 0.080 uDuP, 0.078 n-pris); wtd is the observed water table depth; z are the model parameters for the treatment-specific water table coefficients (-0.008 DP, -0.013 uDP, -0.018 uDuP, -0.043 n-pris); i is the treatment effect (1.171 DP, 1.214 uDP, 1.887 uDuP, 3.535 n-pris)

4 Results

4.1 Precipitation and temperature

The climate at the site is cool and wet (Fig. 1a and b) with large inter-annual differences in precipitation between years. Annual precipitation was 28 % higher in 2008 (1672 mm) than 2009 (1311 mm) and these were higher than earlier reported annual values at WFM of 1140 to 1270 mm (Lees, 1972), but much lower than 2213 mm recorded in 1992 (Jackson et al., 1999).

There was a similar seasonal pattern in soil temperature between the two years of this study, with a higher temperature between May and September (Figs. 1b and 2a).

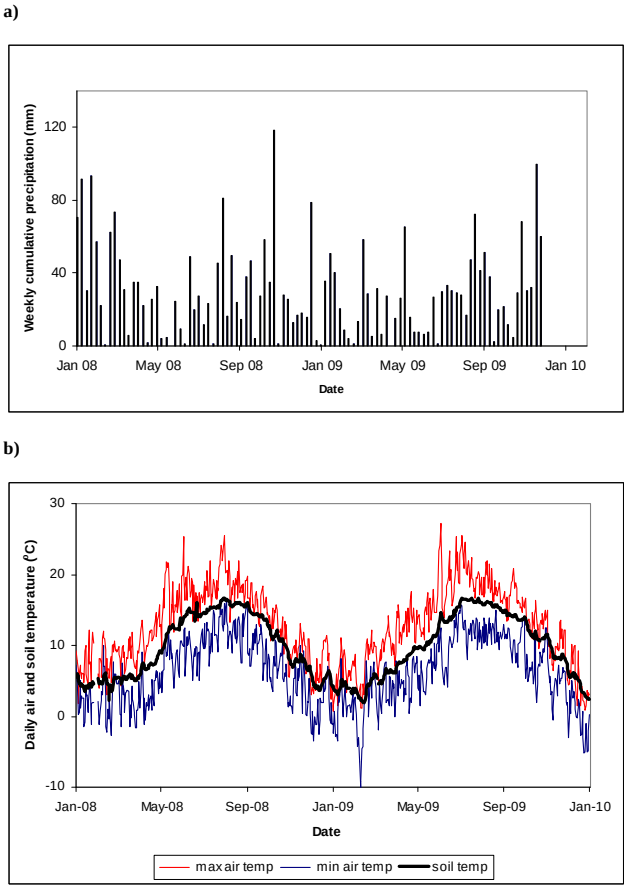

Fig. 1. Environmental variables obtained from meteorological stations near Flanders Moss: **(a)** weekly cumulative precipitation obtained from Auchentroig Estate; **(b)** daily air and soil temperature (30 cm depth) obtained from Portnellan farm.

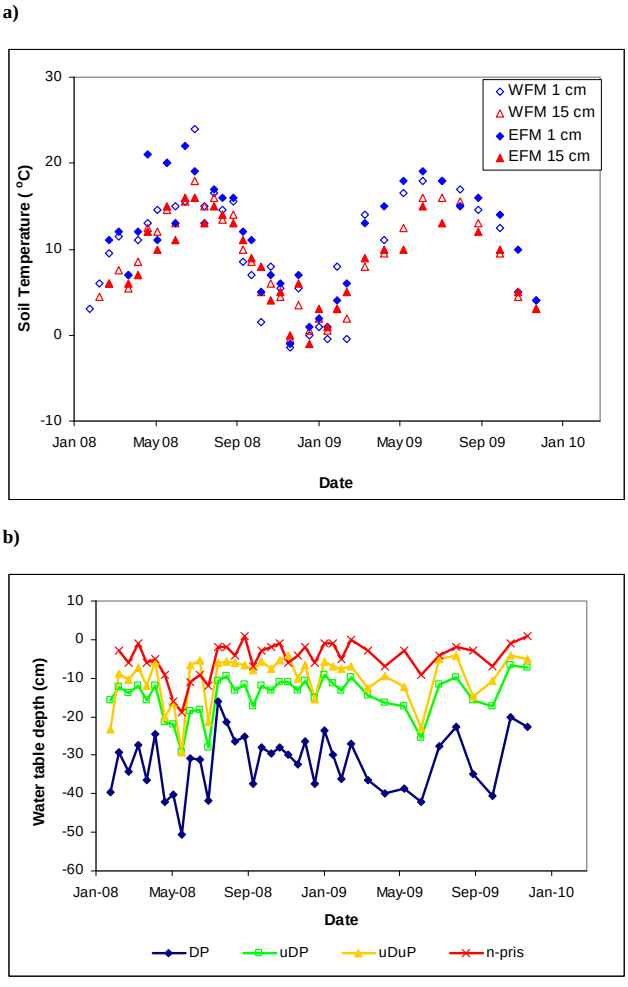

Fig. 2. Environmental variables measured at Flanders Moss during each sampling day: **(a)** soil temperature measured from WFM and EFM (n-pris) sites; **(b)** water table depth from each treatment.

The mean annual minimum and maximum air temperature obtained from the daily meteorological station recording (Fig. 1b) were 6.2 and 12.7 °C, respectively (mean 9.4 °C). Daytime soil temperature measured manually on sampling days at the 1, 5 and 15 cm depth (Fig. 2a) showed no significant differences between the WFM and EFM sites with mean temperatures of 10.7, 8.9 and 8.7 °C, respectively.

4.2 Water table depth and chemistry

The cumulative annual water table depth at the n-pris site for the whole study period was much higher ($p = 0.017$) than the treatments at WFM (Fig. 2b). Within the latter treatments the water depth decreased, as expected, significantly ($p < 0.001$) in the order uDuP > uDP > DP. There were no significant variations between the years and no significant interaction between treatments and years suggesting that the drainage system at Flanders Moss site may have reached a stable condition. The DP treatment had a much lower water table than the other treatments (Table 1) because of the combined effects of the trees and drainage in removing water. There was no clear seasonal pattern in the water table depth although maxima and minima, respectively, reflected high and low periods of precipitation (Fig. 1a).

Water sample analysis of DOC, DON (Fig. 3 and Table 1) and pH (Table 1) from the dipwells showed large seasonal variations with maximum concentrations occurring between late August and early September. The temporal variation followed that observed for temperature, but peak maxima occurred approximately one month later in both years. DOC concentrations were significantly ($p < 0.05$) higher in the uDP treatment than DP and uDuP; the DON concentrations were higher ($p < 0.01$) in uDP than uDuP and n-pris and in contrast, the water pH was lower ($p < 0.05$) in the uDP treatment than the uDuP (Table 1). There were no significant differences between the treatments in the DOC : DON ratio which was very variable (Table 1).

Table 1. Water table depth (WTD) from the soil surface and water chemistry mean values (and ranges) measured across the treatments at Flanders Moss over the duration of the experiment in 2008 and 2009. Different letters within each variable indicate significant differences ($p < 0.05$).

Treatment	WTD (cm)	DOC (mg L^{-1})	DON (mg L^{-1})	pH	DOC : DON
DP	−31.9 (−16.0 to −50.5)^{a}	44.3 (21.7–75.3)^{b}	2.1 (1.0–3.3)$^{a, b}$	4.1 (3.9–4.6)$^{a, b}$	22.4 (13.8–36.1)^{a}
uDP	−14.6 (−6.6 to −30.6)^{b}	59.7 (24.5-108.9)^{a}	2.3 (1.0–3.4)^{a}	3.9 (3.8–4.2)^{b}	28.1 (14.8–43.3)^{a}
uDuP	−9.7 (−4.0 to −29.3)^{c}	36.1 (10.3–87.7)^{b}	1.5 (0.6–2.8)^{c}	4.3 (4.1–4.5)^{a}	25.6 (8.1–48.4)^{a}
n-pris	−4.3 (1.0 to −19.0)^{d}	47.0 (25.8–73.9)$^{a, b}$	1.5 (0.7–3.1)$^{b, c}$	4.2 (4.1–4.6)$^{a, b}$	33.9 (18.3–50.1)^{a}

DP is drained and planted treatment; uDP is undrained and planted treatment; uDuP is undrained and unplanted treatment; and n-pris is the near pristine treatment.

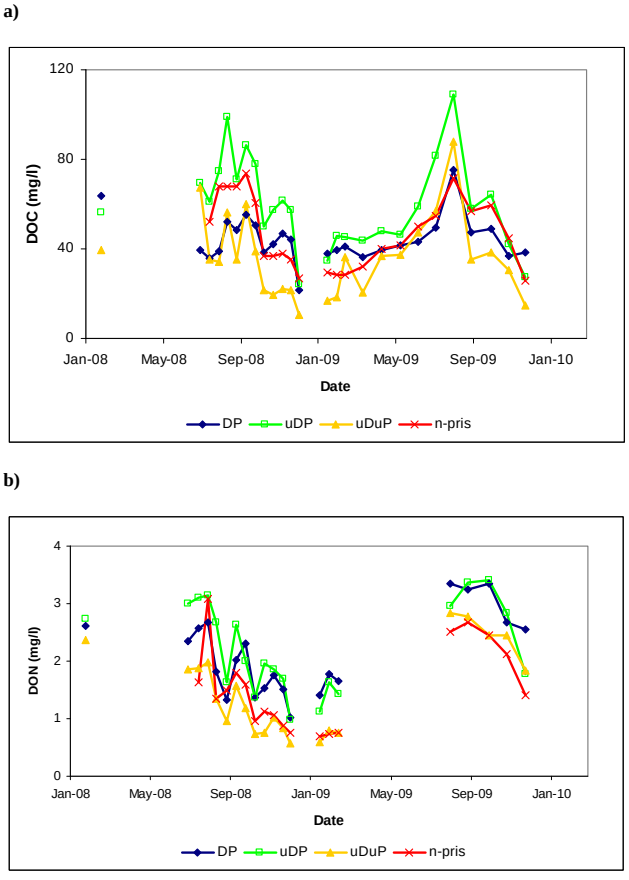

Fig. 3. DON and DOC concentrations measured in the dipwell from each treatment at Flanders Moss. Data missing denotes either not measured or not analysed.

4.3 Soil peat properties

The percentage of C and N, C : N and pH measured in the top 0–10 and 10–20 cm soil layers did not show clear differences between treatments and between peat depths (Table 2). The n-pris treatment, however, showed slightly lower % N (mean of both peat layers, 1.3 %) compared with the mean of those from all the other treatments (1.7 % and 1.4 %). This was reflected in higher corresponding peat C : N ratio values in n-pris treatment (39.2 %) compared with the other treatments (mean 30.3 %). The total C stock in 0–20 cm soil layer, calculated from % C and bulk density for each soil layer, reflected that of the bulk density reducing in the order DP > uDP > uDuP > n-pris.

4.4 Gaseous fluxes of CO_2 CH_4 and N_2O

Fluxes were measured over a total of 365 days in year 1 between February 2008 and February 2009. In the second year, however, the fluxes were only measured over 293 days up to December 2009 after which the measurements were stopped because heavy snowfall made it impossible to locate the chamber frames. Despite the low soil (4.2 °C) and air (3.9 °C) temperatures during this period (December to February) in year one, the cumulative flux calculated for this period alone was ca. 10 % of the annual cumulative flux. Therefore, for each individual treatment, the cumulative flux for year two was extrapolated by factors based on year one fluxes for that period. This did not make any significant difference to the outcome of the statistical analysis so the results for both years are discussed based on a complete 365 day annual period for comparisons. Comparing the mean flux of the 3 or 4 replicates to the median flux for each gas and across all treatments and blocks resulted in a mean flux estimate up to 8 % higher compared to the median value. This relatively small difference (in comparison to flux differences between treatments) is the result of a positively skewed distribution of flux values. ANOVA analyses gave very similar results for both mean and median estimates, but the model distribution and normality of errors were slightly better for the model fit for the median flux as the latter does not include the occasional "spikes" in measured fluxes and, therefore, all analyses were based on the median values.

4.4.1 CO_2

There was a clear seasonal variation in CO_2 emission from all the treatments during both years of monitoring, with 4–5 fold higher emissions during summer months (between May to September) than winter (Fig. 4a). Maximum CO_2 emissions of approximately 21 g CO_2 m^{-2} d^{-1} were measured from the

Table 2. Major top peat layer characteristics of the study site at Flanders Moss.

Treatment	Peat depth (cm)	Total C (%)	Total N (%)	C/N	pH (H_2O)	Bulk Density ($g\,cm^{-3}$)	C stock ($kg\,C\,m^{-2}$)
DP	0–10	50.65 ± 0.61	1.72 ± 0.10	29.78 ± 1.63	3.61 ± 0.08	0.15 ± 0.01	7.83
	10–20	50.81 ± 0.86	1.47 ± 0.07	34.73 ± 1.23	3.60 ± 0.10	0.13 ± 0.01	6.55
	total						14.38
uDP	0–10	49.94 ± 0.46	1.66 ± 0.04	30.18 ± 0.55	3.61 ± 0.06	0.13 ± 0.01	6.49
	10–20	50.82 ± 0.58	1.58 ± 0.06	32.28 ± 1.46	3.64 ± 0.05	0.11 ± 0.01	5.63
	total						12.12
uDuP	0–10	47.81 ± 1.21	1.80 ± 0.08	26.68 ± 1.13	3.77 ± 0.04	0.11 ± 0.01	5.29
	10–20	49.02 ± 0.18	1.74 ± 0.09	28.34 ± 1.28	3.79 ± 0.03	0.11 ± 0.00	5.19
	total						10.48
n-pris	0–10	47.85	1.53	31.58	3.63	0.09	4.08
	10–20	50.59	1.09	46.82	3.57	0.08	3.87
	total						79.5

$\pm$ is the standard error of the mean values.
DP is drained and planted treatment; uDP is undrained and planted treatment; uDuP is undrained and unplanted treatment; and n-pris is the near pristine treatment.

uDuP treatment during mid-summer of both years and efflux patterns for CO_2 from all the different treatments followed that of ambient and soil temperature (Figs. 1b and 2a). Statistical analysis showed that CO_2 emissions were significantly related to the treatments ($p = 0.001$), soil temperature ($p < 0.001$), DOC/DON ratio ($p = 0.008$) and pH ($p = 0.022$). The exponential relationships between CO_2 effluxes from the different treatments and temperature is evident in Fig. 5; the DP treatment had a higher response to temperature than the uDP, probably because of the lower water table and improved aeration in the DP treatment. The highest temperature sensitivity was from the uDuP treatment presumably because of the respiration of substantial ground vegetation present. No significant correlation was observed between the efflux rates from treatments and accumulated total prior rainfall over 24, 48, 72 or 120 h. Annual CO_2 fluxes measured from the different treatments at the WFM site in each year (Table 3) reduced significantly ($p = 0.001$) in the order uDuP > DP > uDP. No significant differences were observed between CO_2 effluxes at WFM and the n-pris site or between year 1 and year 2 annual flux totals (Table 3).

4.4.2 CH_4

Methane emissions (Fig. 4b) showed similar seasonal variations to that observed for CO_2, but peak emissions occurred later than that of CO_2 in year 1 by approximately 1 month. Methane emissions were much higher from the n-pris site compared with the other treatments and from the uDuP compared with those from the uDP and DP treatments with maximum emissions of 197.9 ± 33.1 and $71.2 \pm 53.1\,mg\,CH_4\,m^{-2}\,d^{-1}$, respectively, observed on 19 August 2009. CH_4 emissions were significantly related to the treatments ($p = 0.005$), soil temperature ($p < 0.001$), DOC ($p < 0.001$), DOC/DON ratio ($p = 0.008$) and water table depth ($p = 0.002$). No significant correlation was observed between the CH_4 flux from the different treatments and the accumulated total rainfall over 24, 48, 72 or 120 h previous to the flux measurements. Annual fluxes measured from the n-pris site were $22.63\,g\,m^{-2}\,yr^{-1}$ (Table 3), significantly larger than the treatments at WFM ($p = 0.008$) which declined significantly ($p < 0.001$) in the order uDuP > uDP > DP. No significant interactions were observed between treatments and year.

4.4.3 N_2O

Due to some analytical problems with the gas chromatography analysis, N_2O fluxes were not measured between March and October 2008 (Fig. 4c), so N_2O results are based on year 2 only. N_2O fluxes were generally low with a maximum flux of $1.2\,mg\,m^{-2}\,d^{-1}$ observed from the DP treatment and the minimum flux of $-0.5\,mg\,m^{-2}\,d^{-1}$ observed from the n-pris site. There were no clear seasonal patterns in the N_2O fluxes from the different treatments and fluxes were not related to any of the measured environmental variables. No significant differences were observed between the annual N_2O fluxes (Table 3) measured from the different treatments.

4.5 Modelling peatland GHG budgets

Results of the mixed model analysis identified distinctly different relationships between the three gases and the set of explanatory environmental variables. For N_2O, no significant relationships were found with any environmental variable. Estimates of observed CO_2 were statistically improved by including parameters for soil temperature and management treatment whilst for CH_4, the inclusion of parameters

Table 3. Annual cumulative fluxes calculated for each treatment at Flanders Moss over the duration of the experiment in 2008 and 2009. Different letters across the different treatments indicate significant differences ($p < 0.05$).

Period	GHG	DP	uDP	uDuP	n-pris
Year 1 (2008)	CO_2 ($kg\,m^{-2}\,yr^{-1}$)	1.61^{a}	1.22^{a}	2.58^{b}	$1.84^{a,\,b}$
	CH_4 ($g\,m^{-2}\,yr^{-1}$)	0.14^{a}	$0.54^{a,\,b}$	$5.89^{b,\,c}$	22.12^{c}
	N_2O ($g\,m^{-2}\,yr^{-1}$)	not measured			
year 2 (2009)	CO_2 ($kg\,m^{-2}\,yr^{-1}$)	1.71^{b}	1.24^{a}	2.52^{c}	$1.81^{a,\,b,\,c}$
	CH_4 ($g\,m^{-2}\,yr^{-1}$)	0.16^{a}	$0.75^{a,\,b}$	$9.50^{b,\,c}$	23.14^{c}
	N_2O ($g\,m^{-2}\,yr^{-1}$)	0.08^{a}	0.07^{a}	0.02^{a}	0.09^{a}
mean both years	CO_2 ($kg\,m^{-2}\,yr^{-1}$)	1.66^{b}	1.23^{a}	2.55^{c}	$1.82^{a,\,b,\,c}$
	CH_4 ($g\,m^{-2}\,yr^{-1}$)	0.15^{a}	0.64^{b}	7.70^{c}	22.63^{d}
	N_2O ($g\,m^{-2}\,yr^{-1}$)	–	–	–	–

DP is drained and planted treatment; uDP is undrained and planted treatment; uDuP is undrained and unplanted treatment; and n-pris is the near pristine treatment.

for treatment and interaction terms for water table depth and treatment and for temperature and treatment significantly improved the model fit (Fig. 7). For CO_2, the best fitting model identified that, for a given soil temperature, CO_2 fluxes would be expected to increase in the order uDP < n-pris < DP < uDuP. For CH_4 fluxes the relationship between management treatment, temperature and water table depth is more complex as management treatment significantly affected the observed water table depth. However, for a fixed soil temperature and an average water table depth value for each management treatment, the fitted model predicts increasing CH_4 emissions in the order DP < uDP < uDuP < n-pris.

The fitted statistical models for CO_2 and CH_4 emissions generally showed good agreement between the measured and modelled values (mean of all replicated blocks) for each treatment (Fig. 7). The model, however, was not able to capture the very high flux values which may have been due to factors other than those used such as surface vegetation and time lag for the response of microbial activities to temperature. Nevertheless, the modelled mean annual fluxes for the different treatments were more than 95 % of those measured for CO_2 and 78 % of measured for CH_4. As the current experiment was designed in replicated randomised blocks with replicated gas flux chambers and monitored for 2-yr, it would be expected that the statistical models would provide a robust method for the application to other peatland sites if key environmental variables such as water table depth, soil temperature and vegetation were observed to have similar ranges to this study.

5 Discussion

5.1 GHG fluxes

There are limited published robust year-long data on GHG flux from afforested cool temperate peatlands, particularly from the UK (Billett et al., 2010; Lindsay, 2010; Birkin et al., 2011; Morison et al., 2012) so this study presents the first analysis of the impact of tree planting and drainage on simultaneous CO_2, CH_4 and N_2O fluxes. The comparison with a nearby nearly-pristine site also permits the exploration of the implication of possible peatland restoration on fluxes.

There was no significant difference in the annual N_2O fluxes between the treatments, which is likely to have been because of the high C : N ratio found in this study, resulting in reduced NH_4 supply by mineralisation for the nitrification processes required for N_2O production. There was a clear pattern in the annual CH_4 fluxes from the different treatments, increasing with a higher water table depth in the order DP > uDP > uDuP > n-pris (Fig. 6) agreeing with many observations in the literature. A recent synthesis and analysis of CH_4 emissions from UK soils (Levy et al., 2012) showed a large range of fluxes between -0.15 to $13.8\,g\,m^{-2}\,yr^{-1}$ and estimated the effect of changes in peatland water table on CH_4 emissions as $0.4\,g\,m^{-2}\,yr^{-1}$ per cm increase in water table height. Such an estimate was not possible in this study as the changes in CH_4 emissions with water table depth were not linear, but showed a threshold (see Fig. 6). Most of the time, CH_4 was emitted from all the treatments in this study, except for a few occasions when CH_4 uptake was observed in the DP and uDP treatments (Fig. 7). Emission, even when water table depth was low, indicates that there was usually high microbial methanogen activity and anaerobic zones within the peat surface layer because of the wet ground state. For CO_2, the annual emissions for treatments planted with trees were 35 % higher when drained than undrained, demonstrating the effect of water table depth and aeration.

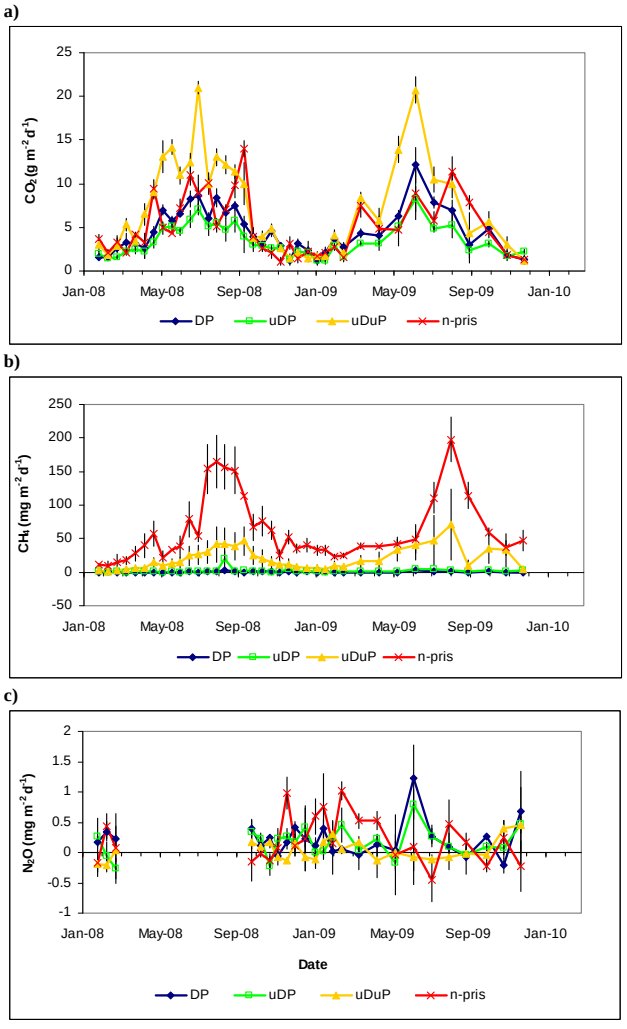

Fig. 4. Fluxes of CO_2, CH_4 and N_2O measured throughout the study period from the different treatments. Error bars for each treatment denote standard error of mean-calculated for replicated blocks. For FME error bars denotes differences between replicated chambers.

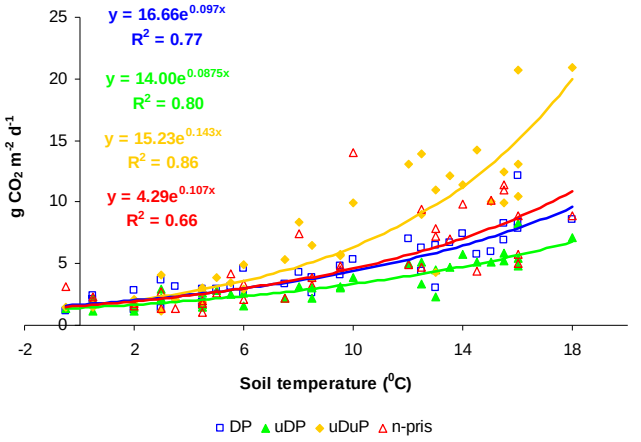

Fig. 5. The exponential relationship between CO_2 effluxes (mean of blocks) and soil temperature measured at 15 cm depth from the different treatments at Flanders Moss between February 2008 and December 2009.

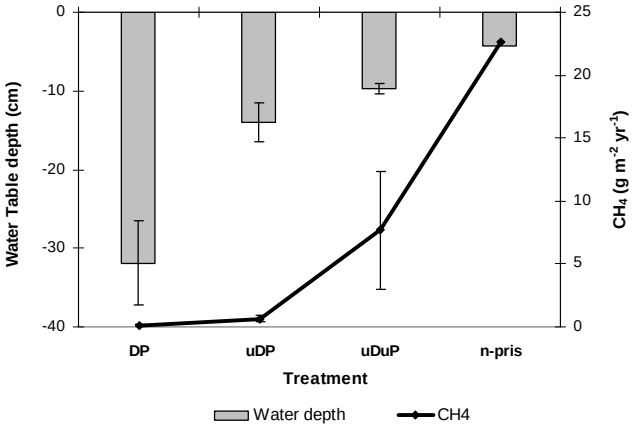

Fig. 6. Mean annual water table depth and CH_4 flux measured from each treatment at Flanders moss during the study period (mean year 1 and year 2). Bars denote standard error of differences between the replicated blocks.

The CO_2 efflux was higher in the uDuP treatment than the planted ones, although the water table was closer to the surface. This was probably because the autotrophic respiration from the substantial ground vegetation cover (even though there were no tree roots) was higher than the heterotrophic CO_2 effluxes from the decomposition of the surface litter below the tree canopy in the planted treatments. Other factors such as lower pH, temperature and litter quality may also reduce the net effect of water table drawdown on CO_2 emission (e.g., Minkkinen et al., 2002). Jungkunst and Fiedler (2007) reviewed the available published annual data to test whether there is a relationship between the global warming potential (GWP) and the water table depth and its dependency on temperature. They indicated that soil moisture is the main determinant of the type of GHG losses, whereas temperature affects the magnitude of GHG emissions both seasonally and regionally. The importance of soil moisture content as a control on soil respiration directly and indirectly through soil temperature was also highlighted by Wickland et al. (2010) from a black spruce forest stand with different drainage classes and by Davidson et al. (1998) from mixed temperate forests.

All the treatments exhibited similar seasonal CO_2 and CH_4 flux patterns (Fig. 4), with generally higher fluxes between May and September, corresponding to the seasonal pattern in soil temperature (Fig. 2a). Most of the treatment differences in CO_2 and CH_4 fluxes occurred during the summer months when fluxes and temperatures were highest, but there were no significant differences in the soil temperature between the treatments. Therefore, temperature alone could not

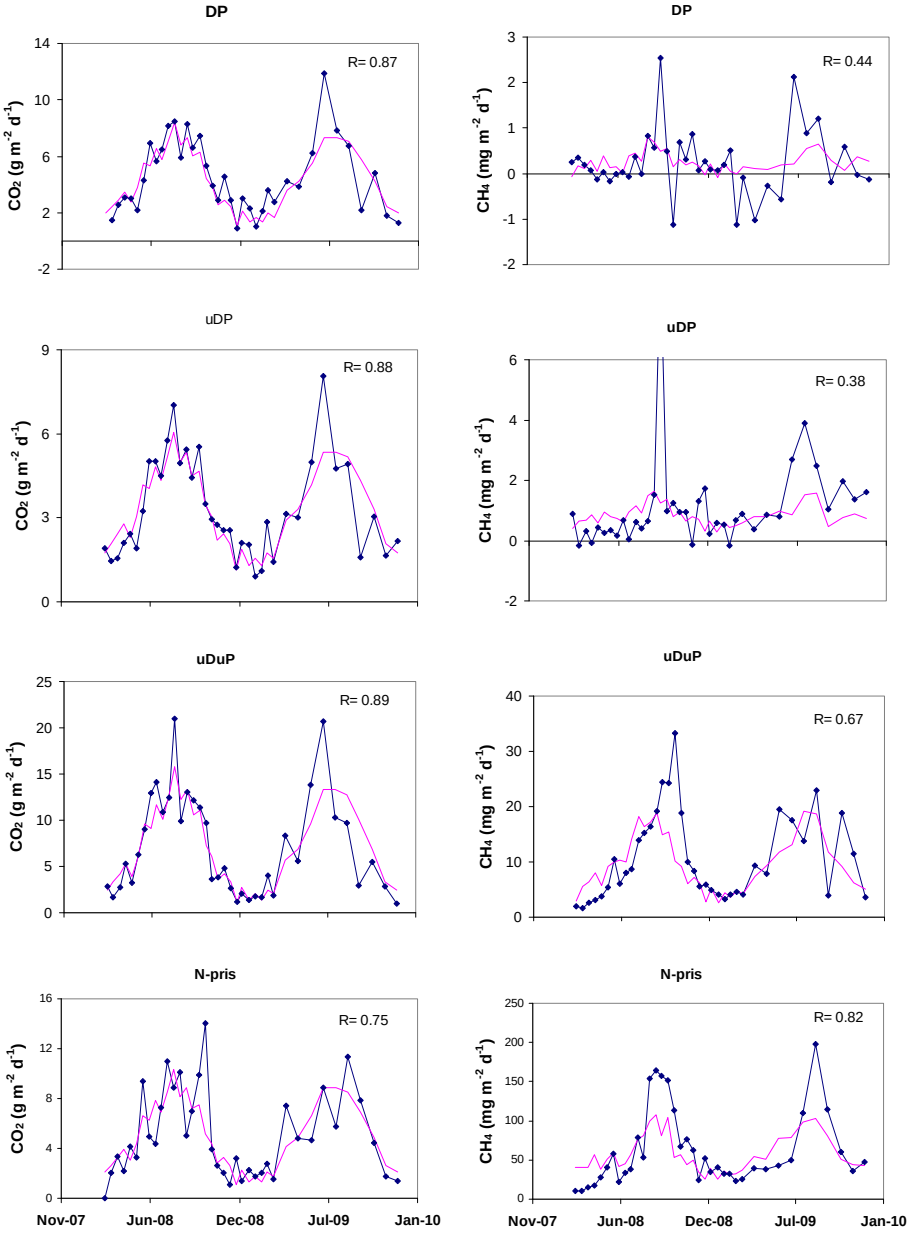

Fig. 7. Modelled (lines) and measured (closed symbol) fluxes of CO_2 and CH_4 for each treatment over the study period. *R* is the correlation between measured and modelled values.

be attributed as the cause of differences in the fluxes between the treatments. This was also evident in all the statistical models fitted to the CO_2 and CH_4 fluxes where significant treatment factor effects were identified regardless of the inclusion of the other informative environmental site variables. This suggests further explanatory variables, which are related to treatment (such as surface vegetation mass and species and tree canopy), also played a key role in the variations between the treatments and need to be identified and used in future modelling of GHG emission variability. Our study supports the conclusion of Dinsmore et al. (2009) that depending on the heterogeneity of the site, flux models could be improved by incorporating a number of spatially distinct sub-models, rather than a single model parameterised using whole-catchment averages.

5.2 Implications for drainage

Drainage and afforestation of peatland affects soil GHG production and consumption processes by lowering the water table depth, enhancing aeration and, thus, increasing decomposition of litter and peat (Clymo, 1984) and nitrogen mineralisation (Freeman et al., 1996). Therefore, it is expected that

drainage and afforestation will decrease CH_4 production (or may even cause net consumption), and they may increase the release of respired CO_2 and sometimes of N_2O emissions. The effect of drainage in lowering the water table and altering GHG emissions was clear in this study. The mean annual water table depth in the uDP treatment was half that in the DP (15 and 32 cm below surface, respectively), which corresponded to four-fold higher CH_4 emissions from the uDP (Table 3). In contrast, the mean CO_2 efflux rate from the uDP treatment was only 74 % of that of the DP treatment.

A recent review of GHG fluxes for UK and European forest soils and for other vegetated sites on deep peat (Morison et al., 2012) reported a wide range of annual fluxes for CO_2 (0.4 to 4.4 kg m^{-2}), CH_4 (−1.0 to 164.0 g m^{-2}) and N_2O (0.0 to 3.0 g m^{-2}). The mean annual fluxes of CO_2, CH_4 and N_2O measured from the different treatments over the study period (Table 3) are within this range, and for the forested DP and uDP treatments the fluxes are close to those measured at sites with similar forest cover and soil. For example, von Arnold et al. (2005a) reported CO_2, CH_4 and N_2O fluxes from drained organic soils with deciduous and coniferous forests in Sweden. Their mean fluxes at a water table depth of 24 cm (similar to the 32 cm depth in the DP treatment here) were 1.44 kg CO_2 m^{-2} yr^{-1}, 0.03 g CH_4 m^{-2} yr^{-1} and 0.08 g N_2O m^{-2} yr^{-1} (Jungkunst and Fiedler, 2007). The CO_2 and N_2O effluxes in this study are similar, but CH_4 efflux was much higher (0.15 g m^{-2} yr^{-1}, Table 3). Significantly higher CH_4 fluxes of 0.63 and 1.75 g m^{-2} yr^{-1} have been measured from drained and undrained sites, respectively, on a peaty gley soils at Harwood Forest in NE England (Mojeremane et al., 2010). However, this could be because those sites were seasonally waterlogged with generally higher water table in their drained and undrained areas (23 and 12 cm depth water table, respectively) than in this study, and the different soil type and peat depth. Nevertheless, they reported 57–76 % decrease in CH_4 emissions in the drained treatment, similar to the reduction by 77 % in this study. Minkkinen et al. (2002) found that CH_4 fluxes from forestry-drained peatland sites in Finland were 50 % lower compared to undrained sites because lowering the water table increased oxygenation, which increased CH_4 consumption. Drainage increased CO_2 emissions in this study by 31 and 38 % in year 1 and 2, respectively. Byrne and Farrell (2005) studied the effect of afforestation on soil CO_2 emissions from drained and undrained ombrotrophic blanket peat in Ireland, afforested 3 to 39 yr previously. They reported in contrast, much lower CO_2 emissions of 0.37–0.95 kg m^{-2} yr^{-1} than in this study from deep peatbog and either lower or similar CO_2 emissions from their drained site compare to the undrained sites. They attributed the differences between the sites to differences in the efficiency of the drainage in lowering water table sufficiently to cause large increase in CO_2 emissions and suggested that their blanket peat sites, despite drainage, are resistant to decomposition.

5.3 Implications for restoration

Restoration of previously afforested peatbogs involves a number of activities and disturbances, such as clear felling, drainage blocking and rewetting. All of which will have a strong effect on the hydrology, soil temperature, vegetation and evapotranspiration of the system. The long-term effect of potential peatbog restoration on GHG fluxes in this study can be estimated from the differences between the annual fluxes measured from the afforested (drained and planted, DP) treatment and those measured from the nearby near-pristine site (n-pris, Table 3), although the effects of the past changes in surrounding land management at that site should be borne in mind. The n-pris site had approximately two orders of magnitude higher annual CH_4 emissions (22.63 g m^{-2} yr^{-1}), compared with the DP (0.15 g m^{-2} yr^{-1}). Although fluxes of CO_2 and N_2O were slightly higher in the n-pris treatment (approximately 35 % and 10 %, respectively), differences were not statistically significant between the n-pris and DP treatments. Dinsmore et al. (2009) measured GHG fluxes from a Scottish ombrotrophic unmanaged peatland (Auchencorth Moss; peat depth ranges from < 0.5 m to > 5 m, and mean annual water table depth of 12.5 cm) on an acid soil with different soil-plant conditions. They reported higher annual CO_2 emissions (3.9 kg m^{-2} yr^{-1}) compared to the n-pris site in this study, but much lower emissions of CH_4 (5.1 g m^{-2} yr^{-1}) and N_2O (0.03 g m^{-2} yr^{-1}). These variations can be attributed to differences in the water table depth and the vegetation cover. The results of our study are at the higher end of the net annual CH_4 flux range of −0.06 to 50.9 g m^{-2} and CO_2 emissions of 0.6 to 2.1 kg m^{-2} reported by Jungkunst and Fiedler (2007) for undrained or restored peatlands in boreal and temperate regions, although higher annual CH_4 emissions of 42.9 g m^{-2} have been measured from an abandoned meadow on peat in the Netherlands (Hendriks et al., 2007).

5.4 Net soil GHG emissions associated with each management

The net soil GHG emissions associated with the drainage or restoration management were calculated using the global warming potential (GWP) of the three GHGs considered here, expressed as CO_2 equivalent (CO_2e), by multiplying the flux of each gas by its GWP over the usual 100 yr time period (1, 25 and 298 for CO_2, CH_4 and N_2O, respectively; IPCC, 2007) and summing (Table 4). CO_2 emissions dominated the net soil GHG emissions associated with the different management treatments contributing 75 % to 98 % of the total GHG fluxes.

Drainage decreased the annual CH_4 emissions by 77 %, equivalent to a net GHG decrease of 12 g CO_2e m^{-2} yr^{-1} (difference in emissions between DP and uDP treatment, Table 4). CO_2 emission increased by only 35 % due to drainage, but this corresponded to a substantially larger net soil GHG of 426 g CO_2e m^{-2} yr^{-1}. This illustrates the conclusion of

Table 4. Estimated net GHG fluxes in $g\,CO_2e\,m^{-2}\,yr^{-1}$, mean of 2008 and 2009, for each treatment at Flanders Moss and change due to drainage and restoration. Values between brackets indicate the standard error of mean of the replicated blocks.

GHG	DP	uDP	uDuP	n-pris	change due to drainage DP-uDP	change due to restoration n-pris-DP
CO_2	1657 (112)	1231 (161)	2553 (122)	1821	426	164
CH_4	4 (2)	16 (6)	192 (117)	566	−12	562
N_2O[a]	22 (10)	20 (2)	5 (6)	26	2	3
total soil GHG emission	1683 (112)	1267 (161)	2750 (169)	2413	416	730
net ecosystem CO_2 exchange	−550[b]	−550[b]		−110 to −420[c]		
net GHG flux	1133	717		1993–2313		860–1180

[a] N_2O is based on year 2009 only.
[b] Calculated from mensuration.
[c] Range from relevant literature.

Jungkunst and Fiedler (2007) on the effect of water table on GHG fluxes that despite the higher GWP of CH_4 it did not outweigh the much larger soil CO_2 losses from soil organic matter decomposition. Equating the difference between DP and n-pris sites as an indicator of the effect of peatbog restoration, suggests that neither soil CO_2 nor N_2O fluxes were significantly affected. Therefore, in contrast to drainage, the net GHG emission change that might be associated with restoration is mainly caused by the large increase in CH_4 emissions of $566\,g\,CO_2e\,m^{-2}\,yr^{-1}$ increasing the net emission by 43 %. Bussell et al. (2010) reviewed the literature to establish how draining and re-wetting of peatland soils can affect GHG fluxes. Their results showed that whilst there was no significant difference in the combined GHG fluxes between drained and undrained peatland, there was a significant reduction in net CH_4 emission of $73\,g\,CO_2e\,m^{-2}\,yr^{-1}$ (range 44 to 102) from the drained, much larger than in this study.

The significance of the contribution of CH_4 and N_2O to the total GHG budget largely depends on the forest and soil type, drainage status and on management practice. The contribution of CH_4 and N_2O to the total GHG emissions for temperate and boreal regions was calculated for a range of restored or undrained peatlands and fens from the data reviewed by Jungkunst and Fiedler (2007) with values up to 65 % for CH_4 and 16 % for N_2O. The effect of restoration inferred from the current study on the total net GHG emission may be underestimated due to additional high emissions expected shortly after restoration disturbances such as clear-felling and drainage (e.g., Skiba et al., 2012; Zerva and Mencuccini, 2005). It is also important to note that the CO_2 efflux estimate is based on soil effluxes only. At the stand-scale, soil CO_2 emissions will be offset by the photosynthetic uptake by trees and other vegetation, so that the contribution of non-CO_2 gases to the net GHG flux will be significantly larger. For our tree-planted treatments, we estimated a total carbon sequestration of $6600\,g\,C\,m^{-2}$, based on the total tree biomass calculated from tree mensuration data for the whole area (Jenkins at al., 2011) which is equivalent to a mean CO_2 uptake rate since planting in 1964 of $550\,g\,CO_2\,m^{-2}\,yr^{-1}$ (although this does not include accumulation of leaf, branch and root litter). Thus, assuming that below canopy biomass increment was minimal compared with that by trees, for the DP and uDP treatments with total soil GHG mean annual net emission of 1683 and $1267\,g\,CO_2e\,m^{-2}\,yr^{-1}$ (Table 4), the net stand-scale emission can be estimated at approx 1133 and $717\,g\,CO_2e\,m^{-2}\,yr^{-1}$, respectively. Note that this calculation is slightly overestimating net CO_2 losses as tree root respiration is included in the measured chamber CO_2 emissions and is implicit in the calculated net tree CO_2 uptake.

Comparing these values with the n-pris site net GHG efflux only of $2413\,g\,CO_2e\,m^{-2}\,yr^{-1}$ implies that the CO_2 uptake by vegetation in this site would have to be larger than 1280 or $1696\,g\,CO_2\,m^{-2}\,yr^{-1}$ in order to have a net negative GHG balance (i.e., sink). Although there is no direct estimate of the vegetation CO_2 uptake by the n-pris site, such a CO_2 uptake is rather large compared to the literature values. For example, the net ecosystem CO_2 exchange from a similar pristine peatland site in Cross Lochs, Forsinard, Sutherland, UK was $370\,g\,m^{-2}\,yr^{-1}$ (Levy et al., 2009) and Billett et al. (2010) suggested that net ecosystem exchange by Auchencorth Moss (mainly grass and sedge with a *Sphagnum* base layer) is between 100 and $420\,g\,CO_2\,m^{-2}\,yr^{-1}$. The review of Lindsay (2010) suggests that the long-term apparent rate of CO_2 accumulation by *Sphagnum* dominated bogs is approximately 110 to $260\,g\,CO_2\,m^{-2}\,yr^{-1}$. Our data indicate that even at the highest net CO_2 uptake rate of $420\,g\,CO_2\,m^{-2}\,yr^{-1}$ indicated by Billett et al. (2010) the n-pris treatment will have a higher net GHG emission of $860\,g\,CO_2e\,m^{-2}\,yr^{-1}$ compared with the drained tree planted site (Table 4) or even higher ($1276\,g\,CO_2e\,m^{-2}\,yr^{-1}$) compared with that of the undrained site.

As climate, soil and forest management factors have different effects on each GHG of interest the only accurate way of quantifying the contribution of each gas to the total GHG budget is by simultaneous monitoring of all gases as in this

study, but also monitoring CO_2 at the stand-level to account for CO_2 photosynthetic uptake.

6 Conclusions

This paper presents the first multi-year measurements of simultaneous soil CO_2, CH_4 and N_2O effluxes from drained and undrained afforested raised peatbog in the UK and the comparison with an adjacent near-pristine peatbog enables the assessment of the potential impact of peatbog restoration on GHG balances. Because of the large scale randomised block design, the well-established (over 40 yr) and replicated experimental treatments, and frequency of gas flux measurements, it enables robust GHG emission factors for these different land managements to be derived.

Fluxes of N_2O were relatively low and no significant differences were observed between the treatments indicating that ombrotrophic deep peatbogs such as in this study are generally low N_2O sources regardless of the drainage and/or restoration status. Temperature variations played a key role in the seasonal variations of CO_2 and CH_4 fluxes, but the differences in the fluxes between the treatments could not be attributed solely to the temperature and/or water table depth. Statistical analyses suggested other explanatory variables, which are related to the management treatments, such as surface vegetation mass, tree canopy and interactions with temperature and water table depth also contributed to the flux differences between the treatments.

Based on soil effluxes, this work shows that drainage (i.e., the difference between the drained and undrained planted treatments) decreased net CH_4 emission by $12\,g\,CO_2e\,m^{-2}\,yr^{-1}$, but increased net soil CO_2 emission by $426\,kg\,CO_2e\,m^{-2}\,yr^{-1}$, resulting in a 33 % higher net GHG emission. This reinforces the case for leaving deep peat areas undrained to preserve soil C stocks. The results here also show that because of the much larger CH_4 effluxes from the near-pristine peatbog site than from the planted drained treatment and the absence of significant difference in soil CO_2 and N_2O fluxes, the net GHG emissions were 43 % higher at the near-pristine site. However, even when likely net CO_2 uptake rates by the peatbog vegetation are taken into account the net GHG emissions of near-pristine peatbog could be significantly larger than the tree planted sites, indicating that restoration of a previously afforested peatland may increase GHG emissions.

Copyright statement

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