Effects of clearfell harvesting on soil CO_2, CH_4 and N_2O fluxes in an upland Sitka spruce stand in England

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Abstract. The effect of clearfell harvesting on soil greenhouse gas (GHG) fluxes of carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) was assessed in a Sitka spruce forest growing on a peaty gley organo-mineral soil in northern England. Fluxes from the soil and litter layer were measured monthly by the closed chamber method and gas chromatography over four years in two mature stands, with one area harvested after the first year. Concurrent measurements of soil temperature and moisture helped to elucidate reasons for the changes in fluxes. In the three years after felling, there was a significant increase in the soil temperature, particularly between June and November (3 to 5 °C higher), and in soil moisture which was 62% higher in the felled area, and these had pronounced effects on the GHG balance, in addition to the removal of the trees and their carbon input to the soil. Annual soil CO_2 effluxes reduced to almost a third in the first year after felling (a drop from 24.0 to 8.9 t CO_2 ha^{-1} yr^{-1}) and half in the second and third year (mean 11.8 t CO_2 ha^{-1} yr^{-1}) compared to before felling, while those from the unfelled area were little changed. Annual effluxes of N_2O more than doubled in the first two years (from 1.0 to 2.3 and 2.5 t CO_2e ha^{-1} yr^{-1}, respectively), although by the third year they were only 20% higher (1.2 t CO_2e ha^{-1} yr^{-1}). CH_4 fluxes changed from a small net uptake of -0.03 t CO_2e ha^{-1} yr^{-1} before felling to a small efflux increasing over the 3 years to 0.34 t CO_2e ha^{-1} yr^{-1}, presumably because of the wetter soil after felling.

Soil CO_2 effluxes dominated the annual net GHG emission when the three gases were compared using their global warming potential (GWP), but N_2O contributed up to 20% of this. This study showed fluxes of CO_2, CH_4 and N_2O responded differently to clearfelling due to the significant changes in soil biotic and abiotic factors and showed large variations between years. This demonstrates the need for multi-year measurements of all GHGs to enable a robust estimate of the effect of the clearfell phase on the GHG balance of managed forests. This is one of a very few multi-year monitoring studies to assess the effect of clearfell harvesting on soil GHG fluxes.

1 Introduction

Forests cover approximately 30% (4.03 billion ha) of the earth's land surface and play a major role in cycling of soil carbon and greenhouse gases (GHG) (Le Quéré et al., 2015). Afforestation and forest management can contribute to GHG net emission mitigation aims by increasing the land-based carbon sink (Grassi et al., 2017). In the UK, 3.2 million ha are covered by forests, 13.2% of the land area (Forest Research, 2020), a substantial proportion of which have been planted over the last 60 to 100 years on peat and peaty gley upland soils (Smith et al., 2018). The UK government has pledged to reach 'net zero' emissions of GHGs by 2050 and to contribute to this goal, the Committee on Climate Change (2019) has recommended afforestation targets of more than 30,000 ha per year. Forest harvesting is an important activity as part of normal forestry practice but also for conversion to other land uses such as part of peatland restoration programmes. Approximately 20,000 ha of trees are felled annually in Great Britain for timber and other harvested wood products (Forestry Commission, 2016).

Forest clearfell harvesting is the phase of the forest management cycle which produces the most disturbance and therefore it is important that it does not impair long-term productivity and benefits. Clearfelling alters (typically increases) many soil factors that influence GHG fluxes. The physical factors include: soil water content and water table height owing to the absence of evapotranspiration from trees (Zerva and Mencuccini, 2005; Wu et al., 2011; Kulmala et al., 2014; Sundqvist

et al., 2014; Korkiakoski et al., 2019); soil temperature through reduced shading (Zerva and Mencuccini, 2005; Wu et al., 2011; Kulmala et al., 2014); soil bulk density, due to soil disturbance and compaction caused particularly by mechanised harvesting equipment (Yoshiro et al., 2008; Mojeremane et al., 2012). Chemical factors include: soil pH (Smolander et al., 1998; Kim, 2008; Kulmala et al., 2014; Sundqvist et al., 2014) and inputs of soil N and organic C (Smolander et al., 2015; Tate et al., 2006; Hyvönen et al., 2012; Kulmala et al., 2014; Sundqvist et al., 2014), due to nutrient release from the decomposition of residual organic matter and root biomass. Interactions between these factors, particularly with the loss of plant litter input and root activity after felling, will strongly affect soil biological processes responsible for production and consumption of GHGs. They will particularly affect nitrous oxide (N_2O) production from aerobic nitrification and anaerobic denitrification, methane (CH_4) production by methanogenic organisms from anaerobic decomposition in oxygen-poor environments or uptake by methanotrophs through oxidation in aerated soils, and carbon dioxide (CO_2) efflux during respiration and decomposition and uptake during photosynthesis. The cessation of the autotrophic respiration (Ra) component of the total soil respiration (Rt) after felling should cause a large decline in CO_2 efflux as a meta-analysis of soil respiration partitioning studies reported that the Ra/Rt ratio in temperate forest soils ranges from 20 to 59% (Subke et al., 2006). In addition the death of tree roots after felling will inhibit the microbial decomposition of root exudates reducing the CO_2 effluxes further. Conversely, the increased soil temperatures following tree removal and higher nutrient availability from decomposition of litter and other plant materials such as brash may increase soil heterotrophic respiration (Rh) (Yashiro et al., 2008).

There are a wide range of results published on the effects of forest clearfelling on soil GHG fluxes. Some studies have shown CO_2 effluxes were reduced (Zerva and Mencuccini, 2005; Korkiakoski, 2019), increased (Tate et al., 2006; Kim, 2008; Kulmala et al., 2014), or unchanged (Butnor et al., 2006; Takakai et al., 2008; Yoshiro et al., 2008). According to Lavoie et al. (2013), the impact is often site specific, affected by the severity of the disturbance or removal of surface organic matter or by the length of time following harvest. Goutal et al. (2012) examined the duration of physical, chemical and biological disturbances in the soil following mechanized harvesting of a beech/oak forest in NE France. Their measurements showed that soil CO_2 effluxes reduced which they attributed to an increase in the frequency and duration of anoxic conditions resulting from poor soil gas diffusion after heavy forestry traffic. Similarly, Kulmala et al. (2014) observed a slight decrease in soil CO_2 efflux in the first growing season after clear-cutting of a boreal Norway spruce stand, although this was probably due to decreased tree root respiration. However, during the following two years, CO_2 efflux at their clear-cut site was significantly higher than in their mature stand, which was attributed to increased decomposition, stimulated by higher soil moisture and temperature. They observed no significant difference in CH_4 uptake due to clear-cutting. However, others have shown that after harvesting forest soils turned from a CH_4 sink to a source (Zerva and Mencuccini, 2005; Sundqvist et al., 2014; Korkiakoski, 2019), or showed reduced CH_4 uptake (Bradford et al., 2000; Wu et al., 2011; Yoshiro et al., 2008), or increased CH_4 uptake (Lavoie et al., 2013). For N_2O, some studies showed increased fluxes (Zerva and Mencuccini, 2005; Yoshiro et al., 2008; Takakai et al., 2008; Ullah et al., 2009; Korkiakoski, 2019), or had no clear change (Tate et al., 2006; Lavoie et al., 2013).

There is therefore a wide range of observed effects, making it difficult to predict the changes in the GHG balance caused by clearfelling in any particular situation, and very few multiple-year studies which characterise the timescale and duration of changes post-felling. Therefore, there is an urgent need to understand and quantify with long-term measurements the effect of forest clearfelling on the GHG budget and to incorporate these into analyses for the complete forest growth cycle for the main forest systems relevant to a country or region (Skiba et al., 2012). The objective of this study was to conduct a relatively long-term (4 year) assessment of the effect of clearfell harvesting on soil (including litter layer) fluxes of CO_2, CH_4 and N_2O in a spruce plantation on organo-mineral soil that is typical of many British upland forests.

2 Materials and methods

2.1 Site description and study layout

The study site was in Harwood Forest, (55°13.1' N, 2° 1' W), Northumberland, north-east England, and comprised a forest area of approximately 4000 ha with an elevation of 200 to 400 m above sea level (Fig. 1). The regional climate is temperate oceanic, with a mean annual rainfall of 1472 mm and mean air temperature of 7.5°C (min. -7 and max. 26.4°C), measured by an automatic weather station (AWS) mounted above the tree stand during the period from April 2015 to April 2018. The tree cover consisted predominantly of even-aged Sitka spruce (*Picea sitchensis,* Bong. Carr.) stands. Sitka spruce is the most common conifer in Great Britain (GB) and represents 26% of the total forested area in GB and 51% of the conifer area (Forest Research, 2020). The main soil type is a seasonally waterlogged organic-rich peaty gley classified as histogleysol with a peat (O Horizon) thickness varying from 15 to 40 cm (Zerva and Mencuccini, 2005), developed in clayey glacial till derived from carboniferous sediments (Pyatt, 1970).

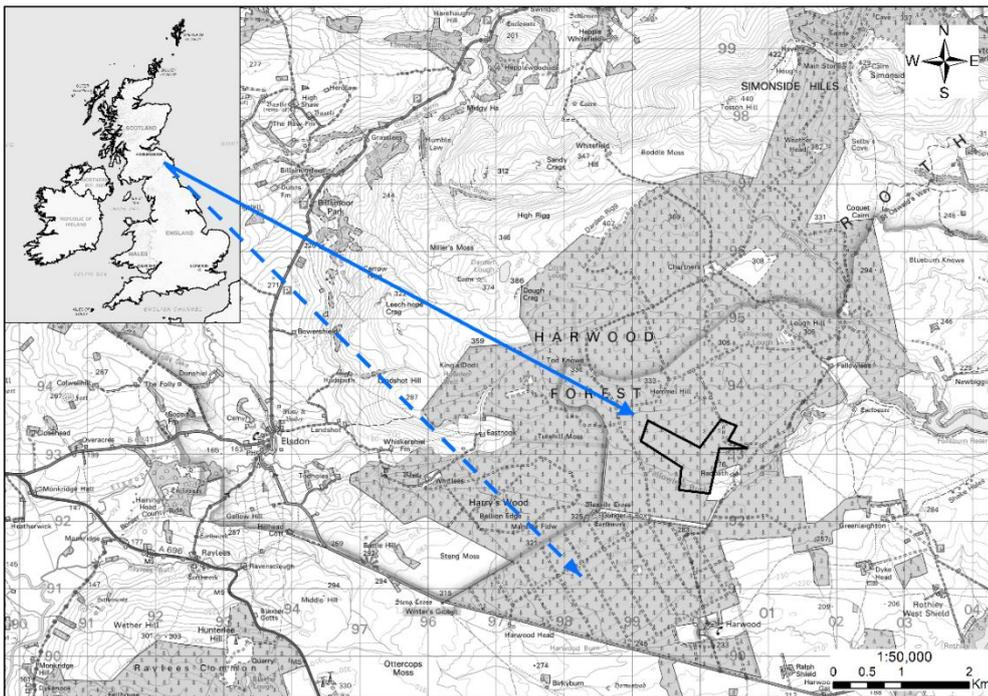



Figure 1. Map of experimental areas at Harwood Forest, Northumberland, UK. The dotted arrow shows the unfelled mature spruce stand area (A), and the solid arrow shows the area that was clearfelled after one year (B), outlined in black. Note: the map is according to © Crown copyright and database right 2021. Ordnance Survey Licence number 100021242.

For the purpose of this study, two nearby areas (A and B, 2 km apart) of mature stands within the forest with similar previous management, soil type and elevation (280 to 290 m) were chosen to carry out measurements over four years between 18 February 2014 to 16 April 2018. During this period, one area (A) was left unfelled (A-yr1 to A-yr4), and the other (B) was clearfelled after one year (B-yr1 before felling and B-yr2 to B-yr4 after felling). In area A, the 40-ha stand was of second rotation, even-aged mature Sitka spruce planted in 1973, with yield class of 18 m^3 ha^{-1} and mean tree density of 1348 trees ha^{-1}. In area B the Sitka spruce stand was planted in 1958, with yield class of 16 m^3 ha^{-1} and mean tree density of 1375 trees ha^{-1} prior to felling, and the felled area covered 42 ha. Felling operations were carried out between late January and early March 2015 and followed standard practices of the Forestry Commission (Murgatroyd and Saunders, 2005). Only timber larger than 7 cm diameter was removed from site, leaving tree tops and branches on site in rows, with some used as brash-mats to prevent compaction of soil by the heavy harvesting machinery.

2.2 Gas flux measurements and analysis

Forest soil fluxes of CO_2, CH_4 and N_2O were measured at approximately monthly intervals over the four-year study period from the two areas of the forest. Flux measurements were made using a modified design of the manual static chamber method described by Yamulki et al. (2013). Each chamber was constructed of opaque PVC with dimensions of 40 cm × 40 cm × 25 cm height to provide a volume of 40 L and placed temporarily on permanently installed frames. The frames were inserted tightly into the ground to a depth of about 3 cm prior to the start of the measurements. The bottom of the chamber had a neoprene rubber foam gasket to ensure a gas-tight seal with the frame, and the top of the chamber had a pressure vent.

Within each area, 8 chambers were positioned randomly in a transect within a 100 m^2 area. During each gas flux measurement, the chambers were placed on top of the frames for up to 60 min and duplicate gas samples of the chamber headspace were taken immediately after closure and then at 3 subsequent 20-min intervals. Gas samples were taken after the chamber was closed by connecting a polypropylene syringe to a 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 within a week using a headspace-sampler (TurboMatrix 110) and gas chromatograph (GC, Clarus 500, PerkinElmer) equipped with an electron capture detector (ECD) for N_2O analysis, a flame ionization detector (FID) for CH_4 analysis, and a catalytic reactor (methanizer) for CO_2 analysis by reducing CO_2 to CH_4 before analysis by the FID detector. The repeatability of the GC gas analysis (assessed as 3 × standard deviation of 20 repeated measurements of standard CO_2, CH_4 and N_2O concentrations at ambient levels) was better than 4% for all gases.

Gas fluxes were calculated based on linear increases of gas concentrations inside the chambers with time. For CO_2 however, if the concentration increase was not linear then fluxes were determined using the R HMR package (version 1.0.1) to plot a best-fit line to the data (Pedersen et al., 2020) to correct for the non-linearity. We did not apply the HMR model for CH_4 and N_2O fluxes as the non-linear fitting plot is very sensitive to variability and outliers in the measured GHG concentrations, particularly for low fluxes and with only four data points per chamber (as also noted by Pihlatie et al., 2013; Brümmer et al., 2017; Korkiakoski et al., 2017) which results in large apparent 'spikes', failure to calculate the fluxes on many occasions, and likely overestimation of calculated GHG fluxes (Pavelka et al., 2018). If CO_2 concentration changes with time were not significant, fluxes for all gases were rejected for that sample as this was judged to be indicative of gas leakage within the chamber headspace. Overall 18 samples were rejected, 9 of which were during snow fall period in January 2016.

Although most studies now measure soil CO_2 fluxes with IRGAs (as noted by Yashiro et al., 2008), which is viewed as more accurate than gas sampling and GC analysis, we needed to use the GC method to measure all 3 GHG within the logistical constraints of the experiment. Therefore, to give further assurance in the gas flux calculations, CO_2 effluxes were compared with those from another 25 static chambers (20 cm diameter, 4.2 L volume, LI-8100-103 Survey Chamber, Li-Cor Inc., Lincoln, Nebraska, USA) measured *in situ* in both areas by a closed loop infra-red gas analyser system (IRGA, LI8100A, Li-Cor Inc.), each over a 2-minute duration after chamber closure (Xenakis et al., 2021). These chambers were positioned over a much wider area than the GC chambers to characterise spatial variation and effluxes were measured for three years but only after felling from Feb 2015. The results (Fig. S1) showed a mean flux difference over the 3-year period of only 6.8% higher by the IRGA method in the unfelled area and 19.5% higher in the felled area compared with the fluxes measured by GC. These are relatively small differences between the two methods, when considering the higher site heterogeneity in the felled area, the inherent differences in the analytical methodologies and as the vegetation in the IRGA chambers was not cut as it was in the GC chambers.

There was no understorey vegetation in the forest stands and no visual evidence of vegetation growth within the chambers in the first two years of the study, with the chamber ground surfaces covered by dense Sitka spruce needle litter. However, in the third year there was growth of Juncus sp. in one chamber in the felled area B between May and September 2016 (maximum height was 40 cm before cutting but the total volume was < 5% of the chamber volume) so the Juncus was cut continually thereafter. The effect of the vegetation cut on the fluxes was assessed from the flux measurements in all the 8

chambers before and directly after first cutting. The statistical analysis revealed no significant differences between mean chamber fluxes before and after cutting for all gases indicating that variations between the chamber fluxes were greater than that due to cutting. In the third and fourth years there was a proliferation of moss in two of the chambers in area A. This could not be removed without substantial disturbance of the soil surface; comparison of chambers with and without moss showed no significant differences between mean fluxes. For CO_2, the effluxes measured will therefore be from aerobic and anaerobic decomposition processes, and respiration of soil organisms and roots.

2.3 Soil moisture and temperature measurements

During each flux sampling day soil temperatures (°C) at 2 cm and 10 cm depths were recorded from one point around each chamber and volumetric moisture content (m^3 m^{-3}) at 6 cm depth was recorded from three points around each chamber. Soil temperature was recorded by a digital temperature probe (Hanna model Checktemp 1) and the volumetric moisture content by a moisture sensor (SM 200 attached to a handheld HH2 moisture meter, Delta-T Devices Ltd, Cambridge, UK). Sensitivity of CO_2 efflux to temperature was determined with a Q10 function (Shi and Jin, 2016) which is the proportional change in respiration resulting from a 10°C increase in temperature, derived from mean daily values of CO_2 efflux for each year using the equation:

$$Q10 = \exp^{10 \cdot b}$$

where b is the slope of the exponential relationship between soil CO_2 effluxes and soil temperature at 2 cm depth, estimated from a log-linear fit. The apparent Q10 values were calculated from the temperature measured near the soil surface (at 2 cm) as recommended by Pavelka et al. (2007), because the strength of the CO_2 efflux and temperature relationship decreases with soil depth.

2.4 Soil parameters and root biomass measurements

To assess changes in soil characteristics due to the felling, additional soil parameters that were likely to have an impact on GHG fluxes were measured between 21 and 23rd July 2015, approximately 5 months after the end of felling. Three replicated soil samples were taken from 0-20 cm depth below the litter layer around each chamber at each site for bulk density, pH, total C and total N content. Soil pH was measured by mixing 5 g soil samples with 25 ml H_2O with analysis using a pH meter probe (Sentek). Soil C and N stocks were measured by the flash combustion method in an NC Soil Analyser (Flash EA, series 1112, Thermo Scientific). Soil bulk density was measured as the mass of oven-dried (at 105 °C until constant weight) soil sample divided by the volume of the cores taken. Live and dead fine root biomass, length and diameter were measured from 3 replicated samples taken from 0-15 cm depth below the litter layer around each chamber in each site. As soil parameters were only measured on one occasion post-felling, the differences between areas could be due to sites, rather than the felling.

2.5 Statistical analysis

Statistical analyses were made using statistical software R (version 3.5.2; R Core Team, 2020). Data were analysed separately for pre-felling and post-felling periods. As measurements were taken from the same eight chambers through time, the analysis was conducted as a repeated measures design, with the significance of felling/non-felling differences determined against the individual chamber data (n=16), not against individual observations. A linear mixed-effects model (nlme, Pinheiro et al, 2018) was used to structure and analyse the data for all flux data. Where necessary, flux data were transformed to obtain a normal distribution. Management type (i.e. felled/unfelled), temperature at 2 and 10 cm and soil moisture (plus all two-way interactions) and date (plus interaction with management type) were treated as fixed effects. Chambers were treated as a random effect (for repeated measures design). For post-felling data, residual analysis indicated that the variance was larger for

190   chambers in the felled area and by individual chamber, therefore weighted variance structures were incorporated within the mixed effects models to account for within-type and within-chamber heterogeneity.

    A range of autoregressive moving average (corARMA , Pinheiro et al, 2018) models were applied to each model, to account for temporal autocorrelation within chambers. Analysis of the (partial) autocorrelation function indicated potential temporal autocorrelation up to one previous time point, therefore all combinations of corARMA structure (up to one previous

195   time point) were applied and the best fit model determined using Akaike's Information Criteria (AIC, R Core Team, 2020) applied to the maximum likelihood fits across each of the gas fluxes separately. The significance of the fixed effects were subsequently determined using analysis of deviance (Anova Chi square tests, Fox and Weisberg, 2011) with non-significant interactions and effects (P>0.05) removed from the final models.

    Annual cumulative fluxes of CH_4, N_2O and CO_2 were estimated to assess the inter-annual variations in each area.

200   Within each year, median flux values were calculated across the eight replicate chambers, along with lower and upper quartiles, maximum and minimum values. These values were then accumulated across the year, by taking the mean of two consecutive flux values and multiplying it by the number of days between the measurements and summing over the monitoring period for each 12 months (as indicated in Table S1) and adjusting to 365 days. For year one where the felling interrupted the measurement in area B, the same cumulative time period was taken for both areas. Annual cumulative fluxes were converted

205   into CO_2 equivalent (CO_2e) using the global warning potential (GWP) for a 100-year time horizon of 34 for CH_4 and 298 for N_2O (IPCC, 2013).

3 Results

3.1 Soil temperature, moisture

Over the 4 years, soil temperature measured during gas sampling varied between 0.1 and 26.0 °C (mean 9.7 °C) at 2 cm soil

210   depth (Fig. 2a) and 1.3° and 15.5 °C (mean 8.3 °C) at 10 cm depth (data not shown), although soil temperatures were never above 15°C under the trees in area A. Before felling in year 1, there were only small differences (p < 0.001) in mean soil temperature between areas A and B (7.1° and 8.5 °C, respectively) at 2 cm soil depth and at 10 cm (7.1° and 7.9 °C) (Table 1), probably caused by sampling the area B later in the day than A. After felling, the mean soil temperature increased in the felled area at 2 cm soil depth (mean 14.3° compared to 9.2 °C in the unfelled area, p<0.001), and at 10 cm depth (10.2 °C

215   compared to 8.4 °C in unfelled), due to removal of the tree cover (Xenakis et al., 2021). In the following two years, the soil temperature remained higher in the felled area by up to 3.4 °C compared to area A, at the 2 cm depth (1.6 °C at 10 cm) and was on average 2 °C higher than before felling (there was no change in the mean soil temperature in A).

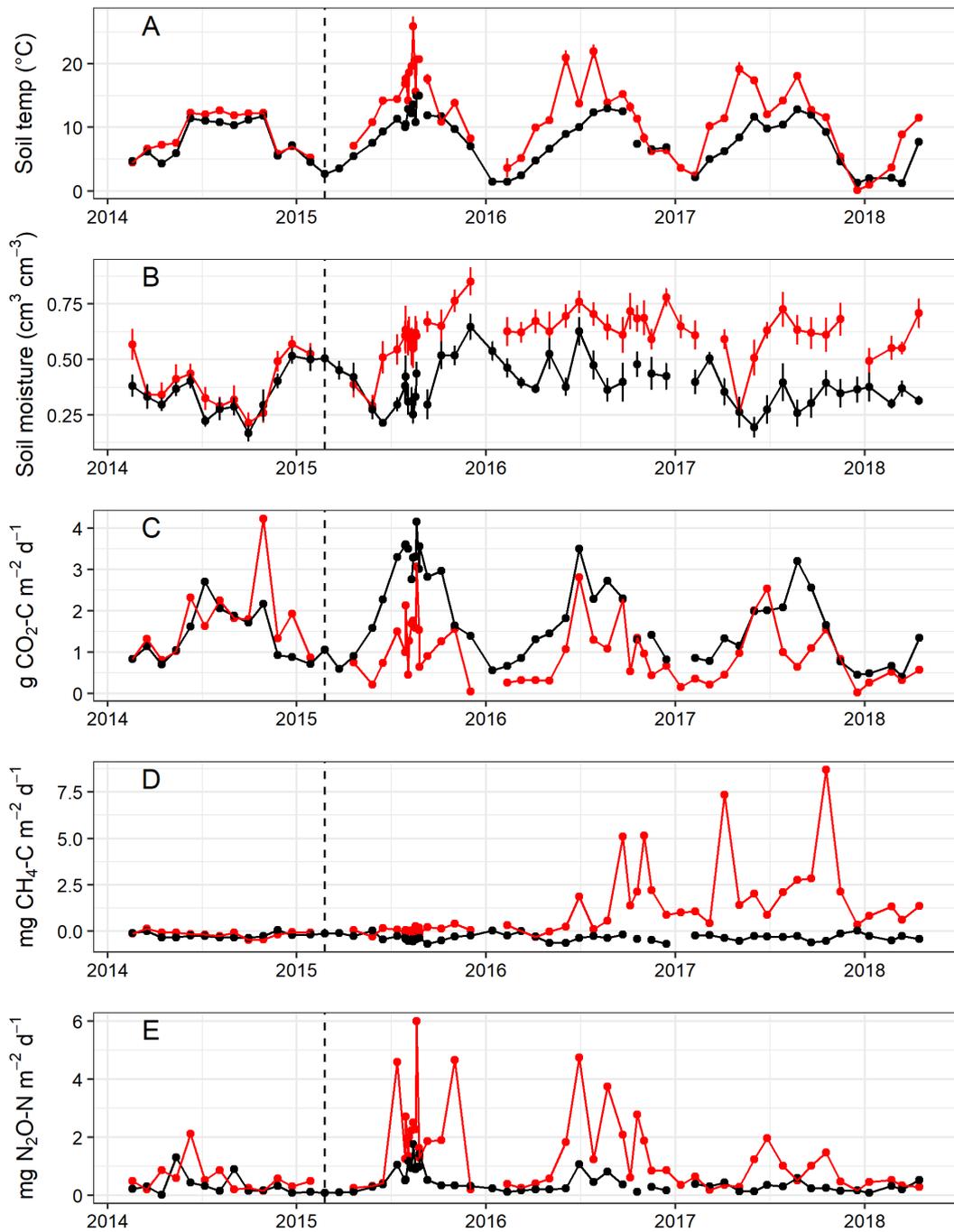



Figure 2. Mean soil temperature (°C) at 2 cm depth (a), volumetric soil moisture content (cm^3 cm^{-3}) (b), and median soil CO_2 (c), CH_4 (d), and N_2O (e) fluxes measured from 8 chambers per area approximately monthly throughout the experiment in Harwood Forest. Black symbols and lines are for area A, (mature spruce stand, remaining) and red are for area B, (mature spruce stand, clearfelled after one year) as indicated by the dotted vertical line. Error bars are standard error of mean of 8 replicate measurements of the soil temperatures and moisture.

Table 1. Annual mean of 8 replicate measurements (±SE) of soil temperatures at 2 cm and 10 cm depths and moisture at 6 cm depth measured close to the soil GHG flux chambers in area A (mature spruce stand, remaining) and area B (a mature spruce stand, clearfelled after year 1) in Harwood Forest. Significance p-values are all < 0.01.

Year	Soil Temp. 2cm °C		Soil Temp. 10cm °C		Soil Moist. $m^3 m^{-3}$	
	A	B	A	B	A	B
1	7.09 (0.10)	8.52 (0.14)	7.12 (0.1)	7.90 (0.13)	0.36 (0.04)	0.39 (0.05)
2	9.22 (0.08)	14.29 (0.52)	8.40 (0.08)	10.21 (0.16)	0.38 (0.05)	0.60 (0.07)
3	7.89 (0.08)	11.25 (0.46)	7.55 (0.06)	8.78 (0.19)	0.44 (0.06)	0.67 (0.06)
4	7.09 (0.09)	10.52 (0.37)	6.88 (0.09)	8.80 (0.21)	0.32 (0.06)	0.58 (0.06)

No significant differences in soil moisture content (by volume) were observed between the two areas before felling with a mean of 0.36 in area A and 0.39 $m^3 m^{-3}$ in B in year 1 (Fig. 2b and Table 1). However, after felling the soil moisture content was significantly higher in the felled area than the unfelled (mean 0.62 $m^3 m^{-3}$ compared with (cf.) 0.38 $m^3 m^{-3}$, p<0.001), due to the reduced evapotranspiration after tree removal (Xenakis et al., 2021). In both areas, there was a pronounced seasonal variation in soil moisture in the year before felling (2014) and in the first few months of 2015, with higher moisture in winter months than in summer, but this pattern was not clear thereafter because the rainfall was more evenly distributed during the last two years of this study (Fig. 3).

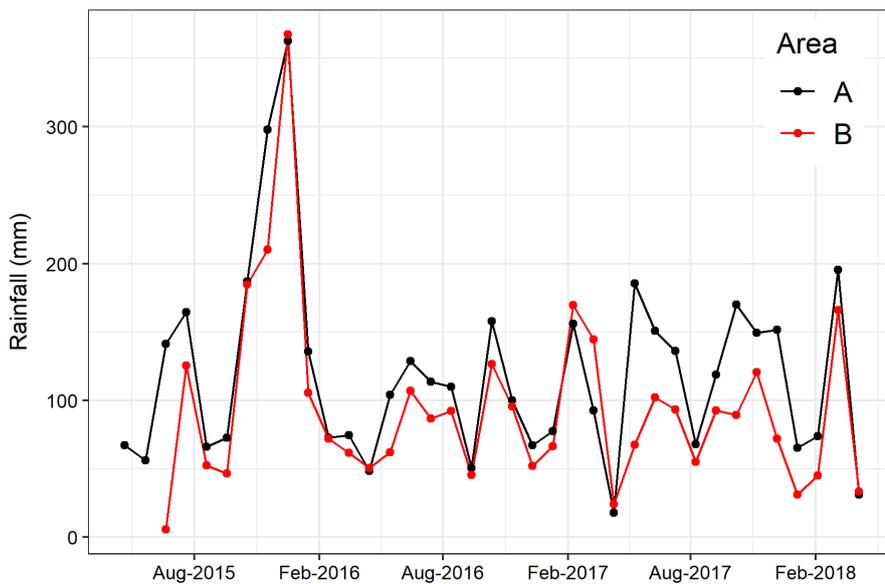



Figure 3. Monthly total rainfall measured after felling in each area in Harwood Forest, atop the 30 m tower in the mature spruce stand (area A, black), and on the ground in the clearfell site (area B, red). Measurements only started in area B on June 2015.

3.2 Soil parameters and root biomass measurements

Soil parameters for the 0-15 cm layer measured 5 months after felling in year 2 showed no significant differences between the unfelled and felled areas in mean soil pH (3.6 and 3.8, respectively) and soil total N content (1.8 and 1.95 %) but the mean soil total C content was about 17% lower (p <0.001) in the felled area (52.0 and 43.3 %). The soil bulk density was significantly (p <0.001) higher at 0.30 g cm^{-3} in the felled area compared with 0.22 g cm^{-3} in the unfelled, but both values are typical for

peaty gley soils (Vanguelova et al., 2013). The higher bulk density and lower C content could be an effect of felling, although as no pre-felling measurements were taken it may be because of site differences. Mean fine (< 2mm) live root mass was much lower in the felled area, as expected (1.6 t ha^{-1} cf. 4.9 t ha^{-1}), but the difference was smaller for the mean fine dead root mass (0.35 and 0.53 t ha^{-1}, respectively), probably due to partial or complete decomposition during the 5-months after felling prior to the measurements.

3.3 GHG fluxes

Large variations were observed in the fluxes between the 8 replicate chambers after felling with some high outliers, particularly for CH_4 (Fig. 2c-e). Therefore, to reduce bias in the annual budget estimates of CO_2, CH_4 and N_2O fluxes and enable a robust comparison between annual fluxes before and after felling, the annual and cumulative fluxes were based on the median of the replicate chambers as described in the statistical analysis section.

3.3.1 CO₂ effluxes

In the first year before felling, there were no significant differences in soil CO_2 effluxes between areas A and B (median 1.54 and 1.75 g CO_2-C m^{-2} d^{-1}, respectively, Fig. 2c). In the following 3 years after felling, CO_2 effluxes became significantly (p<0.001, Table 2) lower in the felled area (median 1.10, 0.90, and 0.92 g CO_2-C m^{-2} d^{-1} in year 2, 3 and 4 respectively) than in the unfelled area (2.44, 1.69 and 1.44 g CO_2-C m^{-2} d^{-1}). There was a clear seasonal variation in the CO_2 effluxes at both areas, which as expected followed that of soil temperature with maximum effluxes during June to September.

Table 2. Results from the analysis of deviance (Chi square tests) showing the probability (p) values for the effects of the explanatory factors/variables: felling, soil temperature (at two depths 2 and 10 cm), volumetric soil moisture measurement date and their interactions (as fixed effects) on soil CO_2, CH_4 and N_2O fluxes in Harwood Forest. Note: p < 0.05 is deemed significant, Pre-fell denotes comparison between area A (mature spruce stand remaining) and area B (before clearfelling) in year 1, Post felling denotes comparisons between the areas for year 2 to 4.

variables	Pre-felling	Post-felling	**Pre-felling**	Post-felling	**Pre-felling**	Post-felling
	CO₂		CH₄		N₂O	
Area	0.352	<0.001	0.803	<0.001	0.588	0.010
Temp.2	1.000	0.587	0.542	0.002	0.269	0.091
Temp.10	<0.001	<0.001	0.436	0.024	<0.001	<0.001
Moisture	0.637	0.850	0.696	0.086	0.002	0.407
Area:Temp.2	0.140	<0.001	0.959	0.574	0.034	0.134
Area:Temp.10	0.200	0.232	0.953	0.547	0.062	0.024
Area:Moisture	0.986	0.600	0.205	0.149	0.031	0.427
Temp.2:Temp.10	0.083	0.170	0.392	0.170	0.967	0.100
Temp.2:Moisture	0.474	0.641	0.882	0.341	0.195	0.192
Temp.10:Moisture	0.346	0.847	0.766	0.605	0.007	0.081

There was no significant correlation between CO_2 effluxes and soil moisture, in part because in the felled area the moisture was high most of the time (Fig. 2b), although low CO_2 effluxes were generally associated with high soil moisture (and lower temperatures) in the winter and the highest effluxes were observed during low soil moisture periods in the warmer temperatures of the summer, particularly in the unfelled area. CO_2 effluxes increased (p ≤ 0.002) with soil temperature at the 2 and 10 cm depths at both areas which was best described by exponential correlation relationships as shown in Fig. 4 for the periods before and after felling. Before felling, the CO_2 efflux response to soil temperature was not significantly different between area A and B, with comparable Q10 values (3.77 and 3.16, respectively, Table 3). However, in the years after felling,

the Q10 values became much lower in the felled area B (mean Q10 = 2.7) than in the unfelled area A (mean Q10= 4.23) with the lowest Q10 value of 2.1 in year 3.

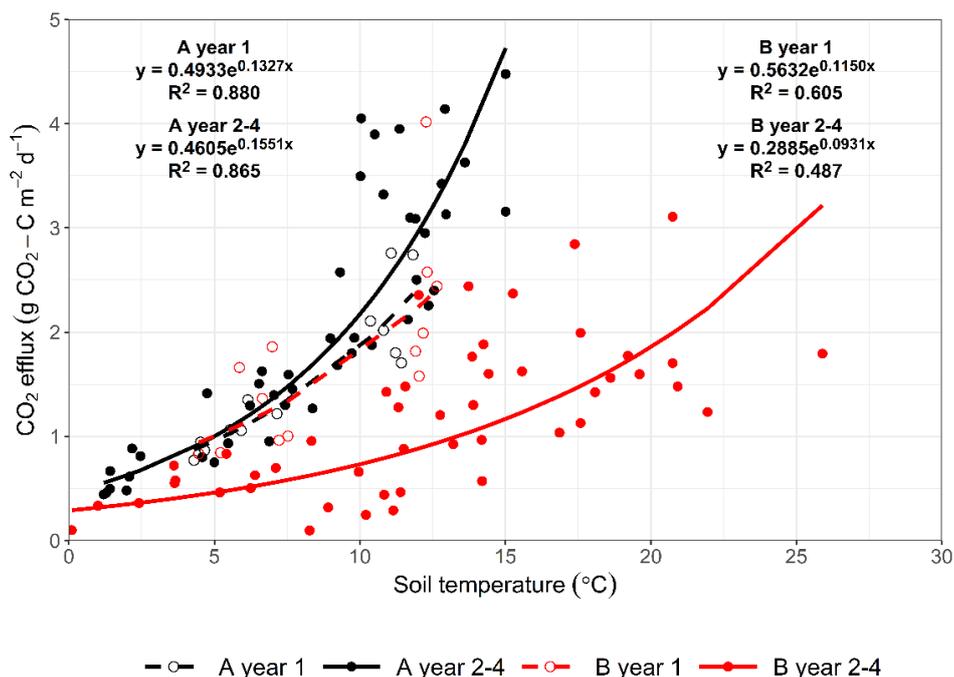



Figure 4. Exponential relationship between soil CO_2 effluxes and soil temperature at 2 cm depth measured from area A (mature spruce stand, black) and area B (clearfell site, red), before (dashed lines, 'yr1') and after (solid lines, 'yr2-4') felling. Equations and R^2 for fitted lines shown.

Table 3. Annual apparent Q10 values for soil CO_2 effluxes in area A (mature spruce stand, remaining) and area B (mature spruce stand, clearfelled after year 1) in Harwood Forest during the study period (as defined in Table S1).

Year	A (unfelled)			B (felled after yr1)		
	R^2	slope	Q10	R^2	slope	Q10
1	0.880	0.133	3.77	0.605	0.115	3.16
2	0.866	0.149	4.44	0.486	0.094	2.57
3	0.700	0.120	3.30	0.319	0.072	2.06
4	0.973	0.160	4.94	0.654	0.124	3.46

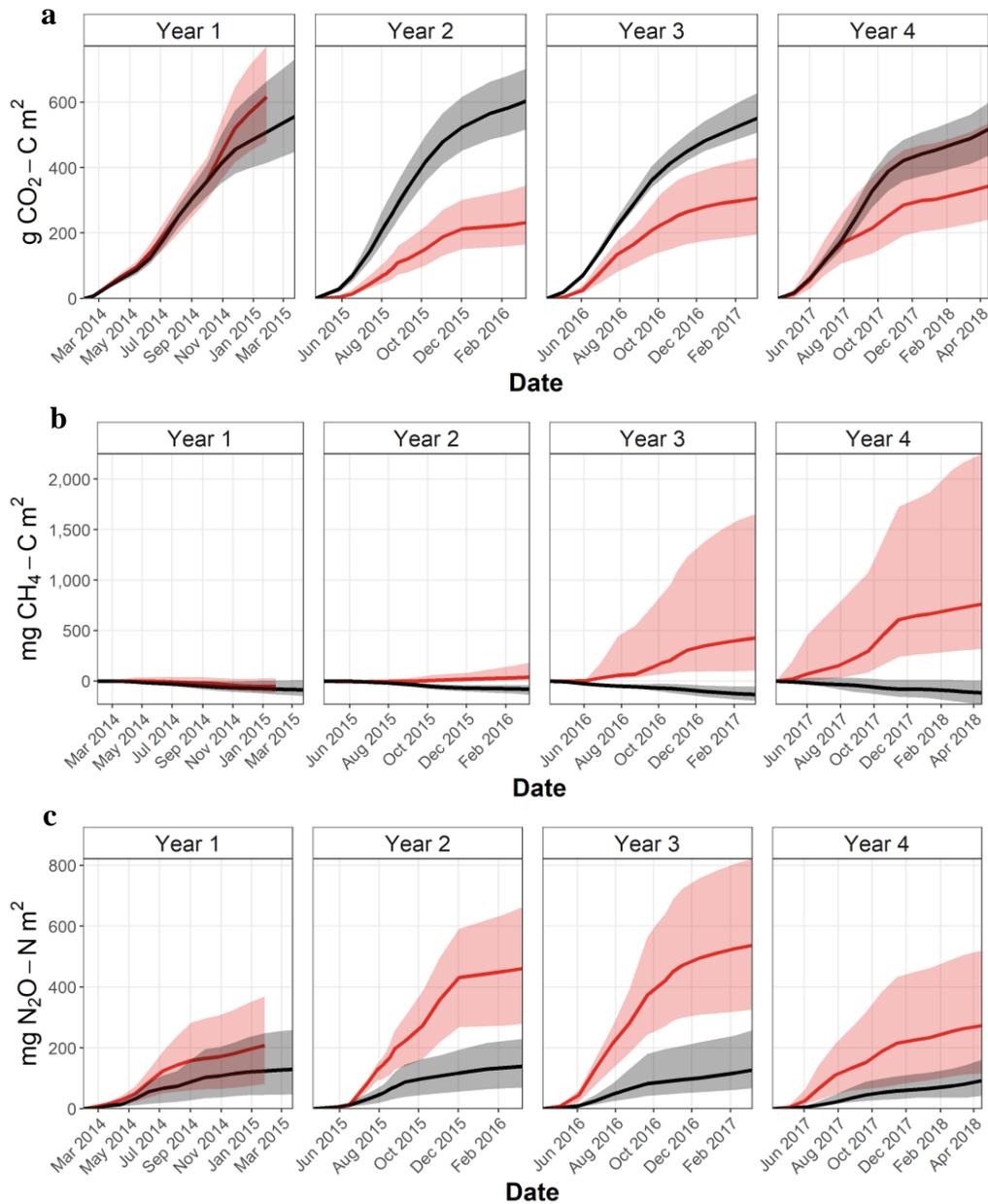
The annual cumulative CO_2 effluxes from the felled and unfelled areas were not significantly different before felling with large overlap between the 95% confidence intervals (Fig. 5a), and the annual CO_2 effluxes were 19.8 and 24.0 t CO_2e ha^{-1} yr^{-1} in areas A and B, respectively (Table 4). After felling, however, there was a clear divergence in the effluxes between the areas, with the CO_2 efflux dropping sharply in the felled area (B). In the first year after felling, the annual CO_2 efflux reduced to its minimum value of 8.9 compared with 23.0 t CO_2e ha^{-1} yr^{-1} from the unfelled area A. In years 3 and 4 the annual effluxes in the felled area increased gradually to 11.3 and 12.2 t CO_2e ha^{-1} yr^{-1}, respectively, but was still lower than the unfelled area (20.3 and 18.4 t CO_2e ha^{-1} yr^{-1}).

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Figure 5. Cumulative soil fluxes of CO_2 (a), CH_4 (b) and N_2O (c) in Harwood Forest from area A (black, mature spruce stand, remaining) and area B (red, mature spruce stand, clearfelled after year 1) during each year, calculated from median fluxes of 8 replicated chambers. Ribbons are the estimated 95% confidence intervals.

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Table 4. Annual soil GHG fluxes expressed as CO_2 equivalents, (t CO_2e ha^{-1} yr^{-1}) using the IPCC Global Warming Potential value for each gas, in area A (mature spruce stand, remaining) and area B (mature spruce stand, clearfelled after year 1) in Harwood Forest. Values in brackets are the % contribution to total CO_2 equivalent.

Year	CO_2		CH_4		N_2O		Total	
	A	B	A	B	A	B	A	B
1	19.82 (97.1)	24.03 (96.0)	-0.038 (-0.2)	-0.026 (-0.1)	0.620 (3.0)	1.034 (4.1)	20.41	25.04
2	23.03 (97.3)	8.87 (79.7)	-0.038 (-0.2)	0.018 (0.2)	0.678 (2.9)	2.246 (20.2)	23.67	11.13
3	20.29 (97.4)	11.30 (80.6)	-0.060 (-0.3)	0.194 (1.4)	0.596 (2.9)	2.524 (18.0)	20.83	14.01
4	18.44 (98.0)	12.24 (88.6)	-0.051 (-0.3)	0.335 (2.4)	0.419 (2.2)	1.243 (9.0)	18.80	13.82

3.3.2 CH₄ fluxes

Soil fluxes of CH_4 in both areas of the forest were generally low throughout the study period, particularly before felling with no significant differences (Fig. 2d). Fluxes were predominantly negative (i.e. removal from the atmosphere) in unfelled area A and before felling in area B with a median flux of -0.33 mg CH_4-C m^{-2} d^{-1} and -0.21 mg CH_4-C m^{-2} d^{-1}, respectively. After felling, area B became a significant (p <0.001) source of CH_4 and fluxes increased rapidly in the following 2 years to its maximum in year 4 (2.48 mg CH_4-C m^{-2} d^{-1}) compared to the unfelled area, which remained unchanged with a small CH_4 sink (-0.33 mg CH_4-C m^{-2} d^{-1}).

Fluxes of CH_4 varied more between the flux chambers after felling particularly in year 3 and 4 (Fig. 5b). Although both soil moisture and CH_4 fluxes increased in the felled area after felling, the increased fluxes and the variation between chambers cannot directly be related to the soil moisture as no significant overall correlation was observed. In addition, including interactions between soil moisture and temperature in the statistical model did not better explain the variation in CH_4 fluxes and difference between areas (Table 2). However, the analysis showed that CH_4 fluxes after felling were best explained by the soil temperature (negative association) at both the 2 cm and 10 cm depths.

Annual fluxes (expressed as CO_2 equivalents using the Global Warming Potential) of CH_4 in the first period before felling were -0.038 and -0.026 t CO_2e ha^{-1} yr^{-1} from areas A and B, respectively (Table 4). In the following years, the felled area (B) became a consistent source of CH_4 of 0.018, 0.194, and 0.335 t CO_2e ha^{-1} yr^{-1} in year 2, 3 and 4 respectively. In contrast, the unfelled area (A) remained a small CH_4 sink with a mean flux value of -0.050 t CO_2e ha^{-1} yr^{-1}.

3.3.3 N₂O fluxes

There were no significant differences in N_2O fluxes between the two areas before felling (Fig. 2e), although the median flux was higher in B (0.33 mg N_2O-N m^{-2} d^{-1} and 0.55 mg N_2O-N m^{-2} d^{-1} for A and B respectively). Maximum N_2O fluxes of 1.30 and 2.12 mg N_2O-N m^{-2} d^{-1} were measured from area A and B, respectively, between May and June 2014. After felling, N_2O fluxes in the felled area B became significantly (p = 0.01) higher in year 2, 3, and 4 (median 1.83, 1.45, and 0.72 mg N_2O-N m^{-2} d^{-1}) than in the unfelled area A (0.63, 0.39, and 0.28 mg N_2O-N m^{-2} d^{-1}) with a maximum flux of 6.01 mg N_2O-N m^{-2} d^{-1} measured in the first year after felling in August 2015.

There were no significant correlations between N_2O fluxes and soil temperature before felling. However, after felling, N_2O fluxes showed a seasonal pattern that followed (p <0.001) that of soil temperature at both depths in both areas with maximum fluxes during periods from June to October. Soil moisture and its interactions with soil temperature (at 10 cm) were the main driver for N_2O fluxes before felling (p = 0.002 and p = 0.007, respectively). After felling, however, the soil moisture remained high (Fig. 2b) and no direct effect on N_2O fluxes was observed. This could be due to the significant (p < 0.01) negative correlation between soil temperature (at both depths) and moisture before felling, compared with after felling where the seasonal pattern in soil moisture was less clear (Fig. 2b and Fig. 3) and no significant correlation occurred, so that the soil temperature became the main driver of N_2O fluxes.

There was a large variation between chambers in N_2O fluxes throughout the study period in both areas of the forest (evident in the confidence intervals shown in Fig. 5c). Before felling, N_2O annual fluxes (expressed as CO_2 equivalents) were 0.62 and 1.03 t CO_2e ha^{-1} yr^{-1} in areas A and B, respectively (Table 4), a much smaller contribution to the total GWP than the CO_2 effluxes. After felling, the annual fluxes of N_2O increased and the highest annual fluxes were measured from the felled area in the two consecutive years after felling with 2.25 and 2.52 t CO_2e ha^{-1} yr^{-1} in year 2 and 3 respectively. However, at the end of the monitoring period in year 4, the annual flux of N_2O returned to a rate similar to that before felling, (1.24 t CO_2e ha^{-1} yr^{-1}).

4 Discussion

4.1 Effect of felling on CO_2 effluxes

Felling and removal of the trees reduced soil CO_2 effluxes by 55%, comparing the mean over the 3 years post-felling with that pre-felling, or 47%, comparing the clearfelled with the mature stand (Table 4). Presumably this was a consequence of the reduction in autotrophic root and rhizosphere respiration (e.g. Boone et al., 1998, Takakai et al., 2008), and measurements about 5 months after felling showed a reduction from 4.9 t ha^{-1} to 1.6 t ha^{-1} in the live fine root mass (i.e. diameter <2mm). Living fine roots and their associated mycorrhizae can contribute up to 59% of total respiration (Ewel et al., 1987; Irvine and Law, 2002; Subke et al., 2006), and for similar spruce stands in Harwood Forest about 40% contribution has been estimated previously (Zerva and Mencuccini, 2005). However, CO_2 effluxes might also increase directly after felling due to an increase in decomposition of fine roots and associated ectomycorrhizal biomass and litter. Therefore, the net CO_2 efflux will be determined by the balance between the reduction in autotrophic root and rhizosphere respiration and the increased decomposition (Köster et al., 2014) which can be short-lived depending on environmental factors such as soil temperature and moisture (Davidson et al., 1998; Skopp et al., 1990).

Before felling and in the unfelled stand, CO_2 effluxes showed a strong seasonal pattern that followed that of soil temperature with higher effluxes during summer, as expected, when fine root density and plant growth activity are highest. The apparent Q10 maximum values of these CO_2 effluxes (Table3) were at the higher end of those reported from temperate deciduous forests in UK (1.60 to 3.92; Yamulki and Morison, 2017 and 2.2; Fenn et al, 2010). The response of CO_2 efflux to temperature became weaker after felling (Fig. 4), even though the soil temperature was substantially higher (a decrease in the apparent Q10 values by up to 36% over the 3-year period; Table 3). This agrees with the study of Zerva and Mencuccini, (2005) who also observed a weaker association of CO_2 effluxes with soil temperature over a 10 month period after felling at another site in this forest. They noted that this was probably because of the increased water content (also evident in our study, Fig. 2b), the death of fine roots and the disturbance of the soil caused by tree harvesting, and suggested that autotrophic root respiration was more responsive to temperature than heterotrophic microbial respiration. However, this effect could also be because the apparent Q10 of soil CO_2 efflux determined from field measurements like these with trees present is influenced by the seasonal changes in radiation and photosynthesis and their positive association with seasonal temperature, rather than an altered temperature sensitivity after felling. A larger reduction of 64% in the Q10 value of soil CO_2 efflux, compared to that found in this study, was reported from a larch forest in eastern Siberia comparing a disturbed clear-cut site with a forest site (2.1 cf. 5.9, Takakai et al., 2008). At the end of our study period in year 4, the Q10 value for soil CO_2 efflux in the felled area (3.46) was close to that of before felling (3.16 in B-yr1), which could be due to ground vegetation growth near but outside the gas flux chambers. It may also be a result of a drop in the rainfall and soil moisture in the period between May and June in year 4 (Fig. 2b, Fig. 3) so the response to soil temperature became stronger, as in the period before felling. There may also have been a recovery from any compaction effects during harvesting, as Epron et al. (2016) showed that compaction by timber forwarding machinery after harvesting a French oak forest on a mineral soil decreased the Q10 values of soil CO_2 by 16-22%. In contrast, Kulmala et al. (2014) found that the temperature dependency of the CO_2 efflux was not affected by clear-cutting of a Norway spruce forest on organo-mineral soil in southern Finland.

The response of CO_2 efflux to soil temperature is likely to have been affected by the substantial increase in soil moisture after felling (Fig. 2b) as noted by Zerva and Mencuccini (2005) and Kulmala et al. (2014). This increase in soil moisture (with values frequently > 0.6 cm^3 cm^{-3}) could have affected soil respiration by limiting the diffusion of substrates and O_2 to microorganisms (Skopp et al., 1990). Yamulki and Morison (2017) could not detect an effect of soil moisture alone on soil respiration in an oak forest in SE England, but the combined model of soil temperature and moisture explained 73% of the CO_2 efflux variations. It is also possible that some of the CO_2 produced may have dissolved in the soil water and gone

undetected (Zerva and Mencuccini, 2005), but probably this effect was negligible here because of the low solubility of CO_2 at the low soil pH (3.8).

4.2 Effect of felling on CH_4 fluxes

Clearfelling changed the soil from a small annual sink of CH_4 to a small net source over the 3-year monitoring period after felling, while the unfelled stand remained a sink in all years. The shift in CH_4 fluxes from net uptake to net emissions by clearfelling has been reported in several other studies: Zerva and Mencuccini (2005) from another site within this spruce forest; Castro et al. (2000) from two slash pine plantations in Florida; Takakai et al. (2008) from a Siberian larch forest soil; Yashiro et al. (2008) from a tropical rain forest in Malaysia; Sundqvist et al. (2014) from a forest site in central Sweden; and more recently by Korkiakoski (2019) from Scots pine nutrient-rich peatland forest in southern Finland. As soil CH_4 production requires anaerobic conditions (Conrad, 2007), this shift from sink to source was probably caused by the substantial increase in soil moisture (62% higher) and soil temperature (6 °C) in the first year after felling. An increase in soil moisture can increase the anaerobic conditions that favour CH_4 production by methanogenic archea (e.g. Sundqvist et al., 2014) and therefore can change the direction of the CH_4 flux. Generally, soil temperature and particularly moisture are considered to be good predictors for CH_4 behaviour (Lavoie et al., 2013) but disentangling the two influences is difficult in field conditions.

In this study, CH_4 fluxes and the flux variations between chambers increased significantly in the following years after felling. The increase in fluxes was modest in the first year after felling but more substantial in the second and third years (Fig. 5b) which may reflect a time lag in the microbial community changing and the fungal decomposition (Glassman et al., 2018). Although the increase in fluxes might be attributed to the substantial increase in the soil moisture, there was no direct correlation between CH_4 fluxes and soil moisture. Some previous studies have also attributed the increase in CH_4 fluxes after felling to an increase in soil moisture or rise in the water table (Sundqvist et al., 2014; Zerva and Mencuccini, 2005; Korkiakoski et al., 2019; Epron et al., 2016) while some observed no direct correlations between CH_4 fluxes and soil moisture after felling (Lavoie et al., 2013; Takakai et al., 2008; Wu et al., 2011; Mäkiranta et al., 2012; Sundqvist et al., 2014 and Zerva and Mencuccini, 2005). The lack of direct correlation between CH_4 fluxes and soil moisture has previously been attributed to: i) insensitivity of methanotrophic activity to small variations in soil moisture and temperature (Sjögersten and Wookey, 2002; Peichl et al., 2010; Wu et al., 2011; Mäkiranta et al., 2012) particularly when fluxes are relatively low such as in this study; ii) CH_4 being produced at a depth greater than that of the measured soil moisture (Zerva and Mencuccini, 2005), and iii) to other overriding biological and physical factors (Lavoie et al., 2013).

Soil temperature determines CH_4 fluxes by influencing methanogenic and methanotrophic activity differently (Luo et al., 2013; Aronson et al., 2013). Generally, CH_4 consumption by methanotrophs is less responsive to temperature than CH_4 production by methanogens as consumption is mainly limited by atmospheric CH_4 diffusion (Dunfield et al., 1993; Kruse et al., 1996). This is in line with the results here, as the statistical analysis showed soil temperature better explained CH_4 fluxes after felling when it became a source than before when the soil was a CH_4 sink. However, soil temperature can be positively related to CH_4 uptake (Maljanen et al., 2003; Wu et al., 2011; Ullah and Moore, 2011; Yamulki and Morison, 2017), emissions (Zerva and Mencuccini, 2005; Dunfield et al., 1993; Ullah and Moore, 2011), or can have no correlation (Takakai et al., 2008, Sjögersten and Wookey, 2009; Lavoie et al., 2013) depending on soil moisture and other factors that affect microbial CH_4 production and consumption. We found no significant interactive effect between temperature and moisture on CH_4 fluxes here. Clearfelling can also affect other factors that play a key role in microbial CH_4 production or oxidation (and CO_2 and N_2O fluxes) by increasing the substrate availability and soil N (NH_{4+} and NO_3^-) concentrations (e.g. Bradford et al., 2000; Wang and Ineson, 2003), for example, as a result of nitrogen release from litter or brash; or reduced soil pH (Dalal and Allen, 2008). The higher bulk density in the felled area (0.30 g m^{-3} cf. 0.22 g m^{-3}) which may have been caused by compaction as a result of the machinery traffic may have contributed to reduced CH_4 uptake, increased CH_4 production and/or CH_4 release (Teepe et al., 2004; Frey et al., 2011). However, the change in soil substrates (organic matter and microorganisms) after felling was not

measured in this study, and the differences between the unfelled and felled area for total soil N content (1.95% and 1.83%, respectively) and soil pH (3.6 and 3.8 respectively) were small.

4.3 Effect of felling on N_2O fluxes

N_2O fluxes (Fig. 2e) increased after felling and there was an association with soil temperature at 10 cm depth (Table 2). Soil moisture was a significant driver for N_2O fluxes before felling, but it became less significant after felling. The effect of clearfelling in increasing N_2O fluxes by similar magnitudes has also been shown from forests on both mineral and peat soils (e.g. Saari et al., 2009; Zerva and Mencuccini, 2005; Mäkiranta et al., 2012; Pearson et al., 2012), but an order of magnitude higher N_2O flux was measured after clearfelling a nutrient-rich drained peatland forest in southern Finland (Korkiakoski et al., 2019).

N_2O fluxes in temperate forest soil are generally expected to be low because of the high C:N ratios in the litter and topsoil (Butterbach-Bahl and Kiese, 2005; Jarvis et al., 2009) but can have a high spatial variability because of the variability in the controlling environmental factors (Peichl et al., 2010; Fest et al., 2009). In this study, both areas of the forest showed large between-chamber variation in N_2O fluxes throughout the study period. This could be due to variations in soil moisture within different flux chambers, particularly after rainfall, and the variability between chambers in soil characteristics, litter amount, mineral N availability and microbial biomass, which were not measured. As the soil moisture was consistently higher after felling (Table 1), the lack of relationship with soil N_2O fluxes may indicate that the moisture content was not limiting for soil N_2O production by the main microbial nitrification and denitrification processes and that other factors were responsible for N_2O flux variations.

As mentioned previously, the total soil N content measured some 5 months after felling was very similar to that in the unfelled area. However, microbial N_2O production in the following years after felling was probably influenced by a declining slow N release into the soil from the decomposition of fresh tree harvest residues and roots (Zerva and Mencuccini, 2005; Yashiro et al., 2008; Saari et al., 2009) but stimulated by warmer soil temperatures in the summer (Kulmala et al., 2014). This is consistent with the significant relationship between N_2O fluxes and soil temperature, particularly at the 10 cm depth, which might imply that N_2O production was more prominent at the deeper, more anaerobic soil depth, probably caused by denitrification brought about by an increased respiratory sink for O_2. In the absence of plants, there is no competition for this newly available N, thereby maximizing substrate availability for microbial N_2O production and release (Skiba et al., 2012). Three-fold higher N_2O emissions from Finnish drained peatland pine forest plots with logging residues than from unlogged have been reported (Mäkiranta et al., 2012). Such effects of increasing soil temperature combined with microbial activities and microbial biomass N in increasing N_2O fluxes have also been reported by other studies (Ineson, et al., 1991; Smolander et al.,1998; Zerva and Mencuccini, 2005; Papen and Butterbach-Bahl, 1999; Smith et al., 2018).

It is pertinent to mention here the study of Liimatainen et al. (2018) from a range of afforested northern peat soils in Finland, Sweden and Iceland, where they suggested that high N_2O fluxes were linked to availability of peat phosphorus (P) and copper (Cu) which could be released with other nutrients by harvesting from soil disturbance and brash (Rodgers et al., 2010), and that low P and Cu concentrations can limit N_2O production even with sufficient N availability.

4.4 Clearfell harvesting effect on the GHG balance

Figure 6 summarises the GHG flux changes by showing the ratio of the annual soil GHG fluxes in B to that in A, with the assumption that the year-to-year variation in the unfelled area A is an indicator equally of what conditions would have been for the mature stand in B if it had not been felled. The figure shows a clear annual GHG flux response to felling where CO_2 effluxes reduced directly after felling, increasing gradually thereafter; CH_4 effluxes increased sharply in year 3 and 4 after felling; and N_2O effluxes increased in year 2 and 3 after felling but decreased thereafter. In order to compare the contributions of each gas to the total GHG soil flux and the effect on emissions and therefore radiative balance, we expressed the fluxes in

CO_2 equivalents as normally done (using their global warming potential values (Table 4). Before felling, the total flux (sum of CO_2, CH_4 and N_2O emissions in t CO_2e ha^{-1}) was 20.4 in A and 25.0 in B areas and CO_2 effluxes dominated the total GWP, contributing up to 97%. The total flux dropped in the first and second year after felling to approximately half (44% and 56% lower annual effluxes, respectively). The contribution of CO_2 to the total GHG flux expressed as CO_2 equivalents in the unfelled area remained constant at about 98% throughout the study period but decreased in the felled area to ca. 80% in the first and second year after felling. This was due to the doubling of N_2O flux which contributed up to 20% of the total flux in the two years after felling. Although the felled site became a source of CH_4, its contribution to the total flux was always small (<2%). For the same periods, the contributions of N_2O and CH_4 to the total flux in the unfelled area were small and remained constant at 2.9% and -0.2% respectively, similar to their values in year 1. In the last year, N_2O annual efflux in the felled area halved to 1.2 t CO_2e ha^{-1}, still 20% higher than before felling, but efflux of CH_4 continued to increase to 0.34 t CO_2e ha^{-1}. Over the 3 years since felling, the total soil GHG flux was reduced by 45% (from 25.0 to 13.8 t CO_2e ha^{-1}) due to the much larger reduction in soil CO_2 efflux than the increases in N_2O and CH_4 fluxes. In the unfelled area there was a reduction of approximately 8%, presumably due to changing weather conditions.
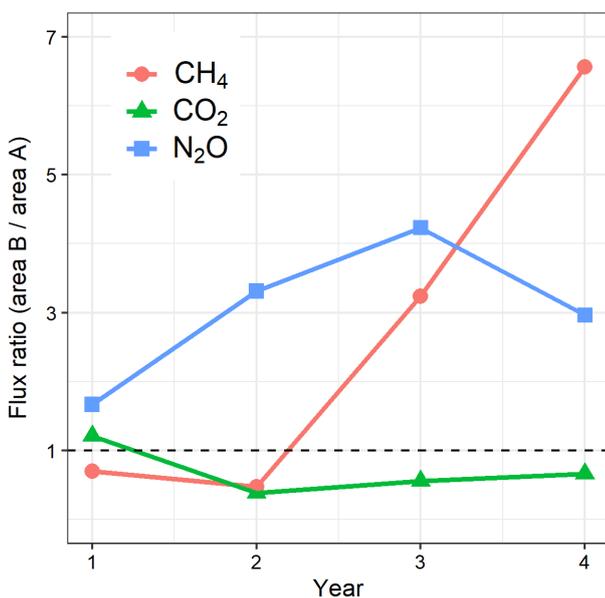
Figure 6. Ratios of annual soil GHG fluxes in area B (mature stand, felled after year 1) to that in area A (mature spruce stand, remaining) in Harwood Forest. Note: CH_4 fluxes presented as absolute values. Dashed line is for ratio = 1 (no difference between areas).

This study is one of very few longer-term assessments of the impacts of clearfell harvesting on the GHG balance of Sitka spruce forest. However, because of the limitations of the periodic manual closed chamber measurement technique used here, it does not take into account the daily temporal flux variations or cover fluxes from ground occupied by brash or stumps. Simultaneous eddy covariance (EC) measurements (Xenakis et al., 2021) over the 3-year period after clearfelling at this study area showed approximately 3 times higher ecosystem respiration (32.8 t CO_2 ha^{-1} yr^{-1}) than our estimated mean annual soil CO_2 efflux (10.8 t CO_2 ha^{-1} yr^{-1}). The difference presumably indicates high CO_2 effluxes coming from the brash mats and stumps, as also suggested by Zerva and Mencuccini (2005), which could not be measured by the small chambers used in this study plus above and belowground respiration of the colonising vegetation which was not included in the chambers. The differences could also be due to spatial heterogeneity over the site as the soil flux chambers are only partially sampling ground that is representative of the EC footprint. Moreover, it has been indicated that peat decomposition after felling is stimulated by

nutrient release from brash (Vanguelova et al., 2010) and therefore higher N_2O and CH_4 fluxes may also be expected as a result of increasing mineral N release and the presence of more labile organic matter, respectively, from areas with brash. Mäkiranta et al. (2012) observed over 3-fold increase in seasonal average (over 3 years between May – Oct) soil N_2O flux and 2-fold in CO_2 efflux from plots with logging residues than without but observed no change in CH_4 emissions. This indicates that brash removal post-harvesting (e.g. for biofuel as suggested by Mäkiranta et al., 2012) might be a way of limiting GHG effluxes from peat decomposition. More information on GHG fluxes from brash and stumps, and the underlying soil processes that might be influenced by felling is a priority for future research.

5 Conclusions

In this upland Sitka spruce plantation on organic-rich peaty gley soil, clearfell harvesting affected soil GHG fluxes by increasing soil temperature and moisture content and reducing fine root mass which affected the soil nutrient and organic C supply and associated microbial populations, activities, and decomposition rates. Although soil moisture increased significantly after felling, there were no direct correlations with the soil GHG fluxes probably because there was limited variation in the high soil moisture after felling. By contrast, there was a good correlation between GHG fluxes and the soil temperature which exhibited much larger temporal variation. This study does not take into account fluxes from brash decomposition, because of the small flux chamber areas, and therefore our total measured soil GHG efflux after felling probably underestimate that of the site as a whole. Over the 3-year measurement period after felling, soil CO_2 effluxes reduced substantially (55%) due to cessation of root respiration outweighing increased decomposition. For the same period, CH_4 fluxes changed from a small net sink to a net source, increasing throughout and N_2O fluxes increased substantially in the 2 years after felling. Mean soil CO_2, CH_4 and N_2O fluxes over the 3-year period after felling contributed 83, 1 and 16%, respectively, to total GHG flux on a CO_2 equivalents basis with an overall reduction of 45%, due to much larger soil CO_2 flux reduction than the combined soil CH_4 and N_2O flux increases.

Author contributions. SY, JILM, GX and MP designed the study. GX, AA and SY carried out the gas sampling and soil environmental measurements. JB carried out the GC analyses, and SY did all the data analyses. JF was responsible for statistical analysis and SY prepared the paper, with contributions from JILM and GX.

Competing interests.   The authors declare that they have no conflicts of interest.

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