Agricultural and Forest Meteorology 295 (2020) 108168

Contents lists available at ScienceDirect

Agricultural and Forest Meteorology

journal homepage: www.elsevier.com/locate/agrformet

Impact of partial harvest on CH_4 and N_2O balances of a drained boreal peatland forest

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ARTICLE INFO

Keywords:
Methane
Nitrous oxide
Flux
Peatland forestry
Selective harvest
Disturbance

ABSTRACT

Rotation forestry including clearcutting is a common method of practising forestry in Fennoscandia. Clearcutting in peatland forests markedly increases environmental loading: leaching of nutrients and methane (CH_4) and nitrous oxide (N_2O) fluxes from soil. Continuous cover forestry has been suggested as an alternative because it does not include clearcutting but partial harvesting. However, impacts of partial harvesting on greenhouse gas fluxes are not well understood and in peatlands have not been studied at all. We conducted a partial harvest by removing 70% of the total stem volume in a mature nutrient-rich peatland forest in Southern Finland. The aim was to investigate how partial harvesting a peatland forest affects CH_4 and N_2O balances, and how much different surface types contribute to the balances. We used automatic and manual chamber methods to measure fluxes from both harvest and uncut control site. Fluxes were measured from the forest floor, logging trails, and ditches. Fluxes from these surface types were upscaled to obtain net ecosystem-level fluxes during two post-harvest summers (June–August 2016 and 2017). After the harvest, forest floor CH_4 fluxes did not change significantly at the harvested site compared to the control site. However, fluxes at logging trails increased significantly. N_2O fluxes increased at the harvest site in the post-harvest years, but so did those at the control site as well. Upscaling CH_4 fluxes to ecosystem-level indicated that despite their small area (2.4%), emissions from ditches could be large on ecosystem-scale, but their uncertainty was high, while the logging trail CH_4 fluxes (20% of the total area) were small. In contrast, N_2O fluxes from ditches were low, but the logging trail fluxes comprised 35–38% of the total surface balance. The overall conclusion is that partial harvesting did not cause considerable changes in CH_4 and N_2O fluxes from a forestry-drained peatland.

Introduction

The most important greenhouse gases (GHG) behind climate change, after carbon dioxide (CO_2), are methane (CH_4) and nitrous oxide (N_2O). Their atmospheric concentrations are much smaller than that of CO_2, but due to their stronger radiative forcing on per-mass, CH_4 and N_2O constitute about 20% and 7% of the global radiative forcing (Myhre et al., 2013), therefore significantly contributing to global climate change. Peatlands store about a third of the global soil carbon (C) pool (Gorham, 1991; Turunen et al., 2002), making them an essential part of global C and nitrogen (N) cycles and contributing significantly to ecosystem-atmosphere exchange of CH_4 and N_2O. They cover ca. 3% of the Earth's land surface (Clarke and Rieley, 2010), and most of them are located in the boreal region (Fischlin et al., 2007).

About 15 million hectares of the boreal peatlands and more than half of the original 10 million ha mire area in Finland has been drained for forestry (Päivänen and Hånell, 2012). Forestry on peatlands, where drainage of the site is one of the most profound measures, affects the conditions for CH_4 and N_2O production significantly. Drainage of peatlands lowers the WTL, which leads to increased oxygen availability in the peat soil above the WTL (Prévost et al., 1997), and thus decreases the CH_4 fluxes from the soil. This is because the CH_4 production ceases in the oxic peat, but also because there is more CH_4 oxidation in the thicker oxic peat layer. In minerotrophic sedge-dominated mires, an even more important factor is the disappearance of sedges following drainage: the input of easily decomposable substrate (labile C) to the

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https://doi.org/10.1016/j.agrformet.2020.108168
Received 18 March 2020; Received in revised form 3 August 2020; Accepted 27 August 2020

anoxic peat layers through sedge roots (Joabsson et al., 1999) ends, and so does the transport of CH_4 from the anoxic layers to the atmosphere through sedge aerenchyma. However, the drainage ditches still emit CH_4 and may negate much of the emission-reducing impact of the drainage (Minkkinen et al., 1997; Minkkinen and Laine, 2006). Drainage typically increases N_2O fluxes only from fertile/nutrient-rich sites (Martikainen et al., 1993). In peatlands, high N_2O fluxes occur mainly from sites with a soil C:N-ratio lower than 25 (Ernfors et al., 2008; Klemedtsson et al., 2005). About 40% of drained peatlands in Finland represent herb-rich and *Vaccinium myrtillys* types (Korhonen et al., 2017), with C:N-ratios typically below 25 (Minkkinen et al. 2020, unpublished results).

Rotation forestry, which is the most common forest management method in Finland and other Nordic countries, includes clearcutting with stem wood harvest at the end of the rotation. Most of the logging residues produced in the harvesting operation are typically placed on the logging trails to create support for the heavy harvesting machinery, and after the harvest, the residues are either collected or left on the site. After clearcutting, the site is typically mounded or harrowed, and the new tree seedlings are planted or seeds sown. In addition to a radical change in hydrological conditions, the soil preparation method after clearcutting influences the CH_4 and N_2O fluxes (Pearson et al., 2012). Clearcutting has been shown to switch forest sites from net CH_4 sinks to sources in both peat (Korkiakoski et al., 2019) and mineral soils (Sundqvist et al., 2014) due to the rise of WTL, although in some studies CH_4 fluxes did not change significantly despite increased WTL (Huttunen et al., 2003; Saari et al., 2009). According to Ojanen et al. (2010, 2013), CH_4 emissions only increase when the WTL is higher than –30 cm. Also, N_2O emissions usually increase after clearcutting (Huttunen et al., 2003; Korkiakoski et al., 2019; Neill et al., 2006; Robertson et al., 1987; Saari et al., 2009), although contrary results with no effect have also been reported (Nieminen, 1998). The reasons behind the increase in N_2O fluxes are unclear, but logging residues have been suggested to enhance N_2O fluxes (Mäkiranta et al., 2012). Clearcutting has also been shown to influence forest micrometeorology by changing surface albedo, net radiation, surface roughness and consequently the energy fluxes (McCaughey & Brintnell 1984; Amiro 2001; Rannik et al. 2002; Kowalski et al. 2003), which indirectly affect CH_4 and N_2O exchange between soil and the atmosphere.

Shifting from rotation forestry towards methods applied in continuous cover forestry (CCF) has been suggested to mitigate the harmful environmental impacts resulting from clearcutting in peatlands (Nieminen et al., 2018). In CCF, only part of the tree stand is removed in cuttings. The cuttings usually include harvesting from above and below (bigger and smaller trees), so that the remaining stand can grow larger and the open places are naturally regenerated. Thus the stand is typically uneven-aged and uneven-structured, it is never clear-cut, and the forestry is not based on stand rotations. The effects of partial harvesting have been studied on various upland forests (Li et al., 2010; Mazza et al., 2019; Sundqvist et al., 2014) but measurements from peatland forests are lacking.

This study investigates how partial harvesting in a mature, drained boreal pine-dominated peatland forest affects the exchange of CH_4 and N_2O between the forest floor and the atmosphere, and what is the contribution of ditches and the logging residues and trails on site-level CH_4 and N_2O balances. WTL raise after partial harvesting is expected to remain smaller than after clearcutting, and therefore, CH_4 emissions should increase less than after clearcutting. Also, N_2O emissions are expected to increase from pre-harvest conditions as mineral N is liberated from decomposing logging residues while fewer trees are taking it up. In this study, the purpose of the partial harvesting was to regenerate the stand naturally by releasing the spruce understorey, leaving most of the birches for contributing to the transpiration potential of the retained stand and as shelter wood for the desired new spruce seedlings expected to show up after harvesting. Should the transpiration potential of the retained stand prove sufficient regarding drainage and, thereby, proper recovery and future growth of the released understorey, ditch network maintenance may not be needed in this connection. Further, with sufficient regeneration of new seedlings, the development towards an uneven-structured, mixed birch-spruce stand could be enabled.

2. Materials and methods

2.1. Site description

The measurements were carried out in a nutrient-rich peatland forest called Lettosuo located in southern Finland (60°38′ N, 23°57′ E). The site was sparsely drained in the 1930s, but proper drainage was carried out in 1969. After the drainage, the site was fertilised with phosphorus and potassium. The distance between the ditches is on average 45 m, and they were dug ca. 1 m deep but have since been partially filled with vegetation. Before partial harvesting in 2016, the two-storeyed tree stand was composed of Scots pine (*Pinus sylvestris*) with some pubescent birch (*Betula pubescens*) in the dominant tree storey with stem numbers of 470 and 190 per ha, dominant heights of 21 and 18.5 m, and standing stem volumes of 184 and 41 $m^3\ ha^{-1}$, respectively. The understorey included mostly Norway spruce (*Picea abies*) (ca. 1000 stems ha^{-1}; 34 $m^3\ ha^{-1}$) and some small-sized pubescent birch (ca. 450 stems ha^{-1}; 7 $m^3\ ha^{-1}$). The tree stand was quite dense, and its irregular shading resulted in patchy and variable ground vegetation. Detailed description of the ground vegetation and soil can be found from (Bhuiyan et al., 2017) and (Korkiakoski et al., 2017, 2019), respectively. At present, the site represents the minerotrophic *Vaccinium myrtillus* site type (Mtkg II) of drained peatlands (for the site type classification in more detail, see Vasander and Laine, 2008).

The partial harvesting (hereafter harvesting) at Lettosuo was performed in 2016 between February 29th and March 16th in an area of 13 ha (Fig. 1a) where all the pine trees (ca. 70% of the total stem volume) were harvested. The logging residues were left at the site, mostly on the logging trails to improve bearing capacity for the harvesting machinery. The trees close to the automatic and manual chambers were harvested without heavy machinery to avoid breaking the measurement setup and instruments. No soil preparation or seedling planting was performed at the harvest site. Also, the ditches were left intact.

A control site of 3.1 ha, with a similar tree stand and vegetation composition (Fig. 1a), located north-east of the harvest site, was left intact. On average, total stem volume on the control site was ca 20% smaller than on the harvest site (245 and 295 $m^3\ ha^{-1}$, respectively); this ratio applied similarly to the dominant pine and the understorey spruce. For birch, the basal areas (7.1; 6.8 $m^2\ ha^{-1}$) and stem volumes (53; 52 $m^3\ ha^{-1}$) were very similar on both sites. When selecting the locations for the intensive study plots (automatic chambers, manual chamber transects), we preferred locations where spruce understoreys were as similar as possible. After harvesting, basal areas of retained spruce around these intensive study plots were 6.8 and 7.4 $m^2\ ha^{-1}$, and stem volumes 31 and 43 $m^3\ ha^{-1}$, in the control and harvest site, respectively.

2.2. Automatic chamber measurements

The automatic chamber system at the harvest site was the same as used in CO_2 flux measurements by Koskinen et al. (2014) and CH_4 flux measurements by Korkiakoski et al. (2017). In addition to the system used by Korkiakoski et al. (2017), we added a N_2O gas analyser in the line (see below). A similar automatic chamber system was installed to both the harvest site (Fig. 1c) and the control site (Fig. 1b); however, two different N_2O gas analysers were deployed at the control site for a shorter period (Fig. S1).

Both chamber systems consisted of six transparent chambers with dimensions 57 cm x 57 cm x 40 cm (length x width x height). The chambers equipped with a fan, photosynthetically active radiation

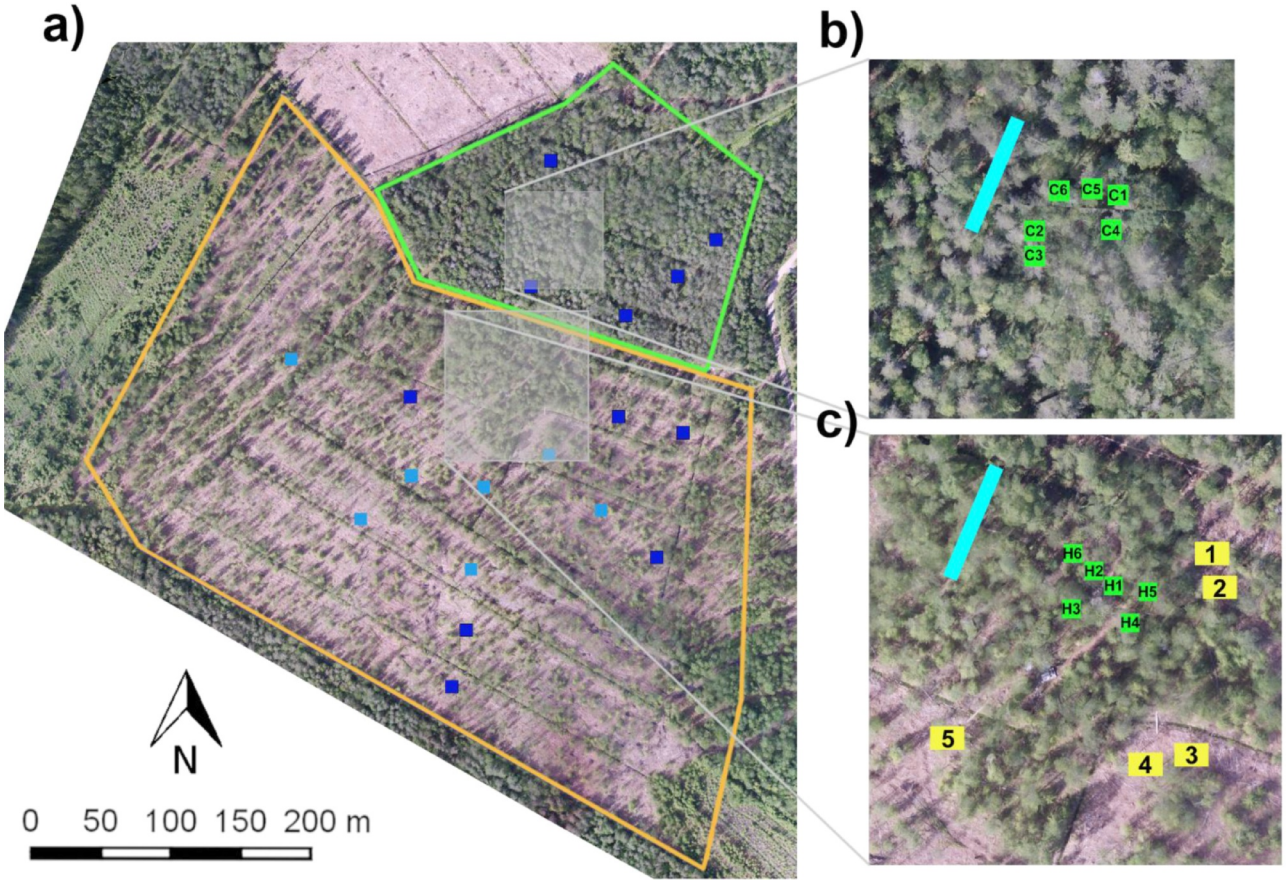

Fig. 1. Aerial view of the site taken in August 2016 (a) and the zoomed view of the chamber measurements at the control (surrounded by the green lines, b) and harvest (surrounded by the orange lines, c) site. The dark blue squares (a) show the automatic WTL measurements started in June 2016 and the light blue squares (a) show the WTL measurements added November 2016. Transect and ditch measurements were located in the cyan rectangles (b, c) and the yellow rectangles (c) show the locations of the logging trail measurements. The automatic chambers were located in the green squares (b, c). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

(PAR; PQS1 PAR Quantum Sensor, Kipp & Zonen, Delft, The Netherlands) and a radiation-shielded temperature sensor were used with permanently installed steel collars (height 5 cm, inserted at a depth of 2 cm) to cover different ground vegetation compositions (Table S1) around ca. 15 m radius from the measurement cabin containing the measurement instruments. During winter 2016–2017, an extension collar (height 16 cm) was placed between the chamber frame and the soil to raise the frame above the snow level. In addition, there was a soil temperature profile at 5 and 30 cm depths (Pt100, PT4T, Nokeval Oy, Nokia, Finland) about 15 m distance to the chambers at both harvest and control sites. Air temperature was measured with HMP45D (Vaisala oyj, Vantaa, Finland) from top of a mast at 25.7 m height, and from 2 m height at harvest and control site. A more detailed description of the chamber system can be found in Koskinen et al. (2014) and Korkiakoski et al. (2017).

A Picarro G1130 cavity ring-down spectroscopy gas analyser (Picarro Inc., Santa Clara, CA, USA) was used to measure the CH_4 and water vapour concentrations with about 4 s interval, and a continuous-wave quantum cascade laser absorption spectrometer (LGR-CW-QCL, Model N2O/CO-23d, Los Gatos Research Inc., Mountain View, CA, USA) was used to measure N_2O concentration with 2 s interval. However, at the control site, Gasmet DX4015 based on Fourier transform infrared spectroscopy (Gasmet Technologies Oy, Helsinki, Finland) was also used in 2017 to measure the N_2O concentration (Fig. S1). The chambers closed once an hour for six minutes. Linear and exponential regressions were applied for flux calculation as described in more detail in Appendix A. Data was filtered with normalised root mean square (NRMSE) and iterative standard deviation methods (Appendix B).

Snow depth inside the automatic and manual (Section 2.3) chambers was measured 1-2 times per month during winter. The precipitation and air pressure data was acquired from the closest official weather station operated by Finnish Meteorological Institute at Jokioinen (~35 km northwest of Lettosuo).

2.3. Manual chamber measurements

2.3.1. Transect and ditch measurements

Manual chamber sampling transects (Fig. 1bc) were set up on harvest and control sites similarly as described in Korkiakoski et al. (2019). Measurements were made between 29 June 2015 and 29 August 2017, mostly during the snow-free periods. The measurement interval mainly varied between one week and one month; however, there were a few longer gaps in autumn 2015 and spring 2016.

To have a spatially representative sampling in relation to the ditches at the harvest and control sites, two parallel flux measurement points were established (located within 2 m from each other) in the ditch and at a distance of 4, 8, 12 and 22.5 m from the ditch (hereafter 'transect'). 2 cm deep grooves were carved into the soil surface for the chambers, and were renewed when necessary to keep the chamber sealing adequate. However, if the soil surface was frozen, the measurements were not made as it was not possible to seal the chamber properly. No aboveground logging residues were introduced into the manual chamber points. For ditch measurements, extra collars (height 30 cm) were installed into the ditch one hour before the measurement.

The fluxes were measured using a closed-chamber system with an

opaque cylindrical chamber (height 30.5 cm, diameter 31.5 cm) including a mixing fan. The analysis of gas concentrations was made either using (1) a portable analyser or (2) inside-cabin laser analysers which were detached from the automatic chambers during the manual measurements. As a portable analyser, we employed the same Gasmet DX4015 (Gasmet Technologies Oy, Helsinki, Finland), which was also used in the automatic chamber system at the control site. The air circulated in a loop between the gas analyser and the chamber with a closure time of 10–11 min. The portable gas analyser was used at the transects in harvest and control sites in 2015 (Fig. S1). However, after the harvest in March 2016, more precise inside-cabin analysers coupled with the automatic chamber system were exploited in manual measurements at both transects. This method allowed capturing lower fluxes, especially CH_4, more precisely with a shorter closure time (5 min). At the control transect, N_2O fluxes were measured with the Gasmet DX4015 connected to the automatic chamber system, with a 10 min chamber closure time.

2.3.2. Logging trail and residue measurements

Five parallel plots with four different types of measurement points were set up on the harvest site (Fig 1c.) at the beginning of May 2016 by installing chamber collars (55 × 55 × 20 cm, length x width x height) 15 cm deep into the soil. The measurements continued until November 2017 (Fig. S1). Two of the points were located on the logging trail where the residues were piled for additional support for the harvesting machinery. The first point was established on the ground compressed by the wheels of the heavy machinery while the second one was set up on the uncompressed ground. The other two points with a small amount and no logging residues were located outside the logging trail. One of these two points simulated a case where trees were harvested, but a small amount of residues were left on the ground while the other one simulated an area which was left untouched during the harvest. Each point had a temperature sensor (iButton DS1921G, Maxim Integrated Products) inserted at 5 cm depth. Also, the points on the logging trail had a similar sensor on the soil surface under the residues. The dry mass of the logging residues in each collar was determined in May 2016 as follows: The residues inside the collars were temporarily removed from the collars and divided into five groups: *Pinus Sylvestris* twigs with needles and branches without needles, *Picea abies* twigs with needles and branches without needles, and *Betula pubescens* branches and twigs. The fresh weight of each of these groups was recorded, and then the residues were returned inside the collars. Also, similar sample groups nearby the collars were taken and weighed and dried at 105 °C. After drying the needles were separated from the twigs, and the dry weight of the sample groups was measured. The relations of these dry and fresh masses were used to determine the dry mass of the needles and woody material (branches and twigs) inside each collar (Table S2). The vegetation composition at the points was recorded on 6 September 2016 (Table S3).

The flux measurements were made by exploiting the instruments used in the automatic chamber system similarly to the transect measurements from 2016 onwards (Sect. 2.3.1) using an angular chamber (60 × 60 × 30 cm, length x width x height).

2.4. WTL measurements and calculations

WTL was monitored hourly with automatic probes (TruTrack WT-HR, Intech Instruments Ltd, Auckland, New Zealand; Odyssey Capacitance Water Level Logger, Dataflow Systems Limited, Christchurch, New Zealand) in dipwells (perforated plastic tubes 120 cm long, 3.5 cm diameter) in multiple locations at both partially harvest and control site. Four dipwells were set up at both automatic chamber systems (Sect. 2.2) next to the chambers in December 2015. When the chambers were close to each other, only one dipwell was installed between them. At the transects, dipwells were installed in May 2015 at each four distance (4, 8, 12, 22.5 m) from the ditch next to the chamber points (Sect. 2.3.1). Also, each plot on the logging trails (Sect. 2.3.2) had one WTL probe since May 2016 until October 2017. In addition to WTL measurements next to the chambers, WTL was also monitored around the control and harvest sites. Five dipwells with automatic WTL probes were installed at the control site (Fig. 1a) in June 2016. Also, at the harvest site, six dipwells were installed in June 2016, and 7 additional dipwells were installed in November 2017 (Fig. 1a). All these WTL measurements were later used in flux upscaling (Sect. 2.5). The WTL measured at the logging trail plots was used to estimate the WTL at each point in the plots by taking into account the ground elevation. These estimated WTLs were used in the mixed-model analysis when investigating the environmental variables affecting the logging trail fluxes (Sect. 2.6).

2.5. Upscaling of the CH_4 and N_2O chamber measurements

To upscale the chamber measurements to the whole harvest and control sites, we used a combination of drone and satellite images to identify the sizes of the different surface types at the site. First, the length of the ditches was measured from the drone images assuming a ditch width of 1 m. Due to lack of any data about the wetness of the ditches, we assumed half of the ditches dry like the ditch measured at the control site, and half wet like the measured ditch at the harvest site. For the harvest site, the ditch area was estimated at 0.315 hectares, which was about 2.4% of the total harvest area (13 ha). For comparison, the share of the ditch area was assumed to be the same for the control site as well. The remaining surface type was forest floor, which at the harvest site was further divided to logging trails and non-logging trails while the whole forest floor at the control site was classified as forest floor without logging trails. The width of the logging trails was 4 m. There were usually two logging trails across one strip (width ~40 m); therefore, the area of the logging trails was assumed 20% of the total forest floor area.

The upscaling was performed for summers (JJA) 2016 and 2017 for harvest and control site separately. The upscaling was limited to summertime only when the measurement interval of all forest floor types was the highest. The method was based on the relationship of daily WTL and CH_4 flux (Fig. S2) and any other possible factors affecting the flux were ignored. However, there was no clear correlation between N_2O flux and environmental variables (data not shown). Therefore, the N_2O flux upscaling was based only on the emission levels of the different surface types and their share of the total area. The ditch fluxes of both gases were gap filled with linear interpolation separately for the ditches located at the harvest and control sites, and later combined by taking a daily mean between the sites. This averaging assumes that 50% of the ditches were wet and 50% were dry and that the harvesting did not affect the ditch fluxes. For the forest floor, daily mean fluxes of automatic, transect and logging trail chamber measurements were binned into WTL bins with a size of 5 cm ranging from –10 to –75 cm. The binning was made separately for logging trails and non-logging trail forest floor. Logging trails included the chamber measurements made on the logging trails while the forest floor measurements included the automatic chamber and transect chamber measurements, but also the logging trail chamber measurements made outside the logging trails (Sect. 2.3.2.). Next, the mean flux for each WTL bin was calculated. Then, control and harvest sites were divided into equally sized plots according to the number of automated WTL measurements located around the site in different compass directions. The areal share of ditches, forest floor with logging trails and forest floor without logging trails were assumed to be the same across the plots (for harvest: 2.4%, 20% and 77.6%). The control site had five automated WTL loggers; therefore, it was divided into five plots, and the harvest site, due to a different amount of WTL dipwells in summers (Sect. 2.4), was divided into 6 and 13 plots in 2016 and 2017, respectively. At each plot, the daily mean CH_4 flux was acquired from the WTL bin corresponding to the daily mean WTL. Finally, the mean fluxes across the plots were

Table 1
Annual and summertime (JJA) mean air temperatures and precipitation, and the annual maximum snow depth during the measurement years (2015 – 2017) and the previous climatological normal period (1981 – 2010; Pirinen et al., 2012).

	Air temperature [°C]		Precipitation [mm]		Maximum snow depth [cm]
	Annual	Summer	Annual	Summer	Annual
1981-2010	4,6	15,2	627	218	28
2015	6,3	14,4	680	208	33
2016	5,3	15,5	536	199	19
2017	5,2	14,1	657	240	12

calculated for each surface type and added together to acquire the CH_4 balance of the whole forest.

The uncertainty of the upscaling was estimated by taking into account the most significant uncertainty sources. The ditch area uncertainty was estimated at 51%, which included both the uncertainty of the ditch area (10%) and the uncertainty of the wetness of the ditches (50%). We did not know how wet the ditches were, which can have a large impact on the CH_4 fluxes. The uncertainties of the forest floor fluxes were estimated from the summertime balance uncertainties (Appendix C) measured by the automatic chambers. The mean uncertainty between the chambers was calculated for the harvest and control site separately for each year. For the ditch and logging trail fluxes, the uncertainty was estimated by dividing the standard error of the mean with the mean summertime flux. The estimated uncertainties and their sizes are shown in Table S4. The errors of the areas and the fluxes were combined with the standard error propagation principle to acquire the total uncertainty of the upscaling.

The flux calculations and data analysis excluding statistical tests mentioned above were made with the Python programming language (Python Software Foundation, version 2.7, https://www.python.org) using NumPy (http://www.numpy.org/), SciPy (http://www.scipy.org/), Pandas (http://pandas.pydata.org/), and matplotlib (http://www.matplotlib.org) libraries. The flux calculation methods for the automatic chambers were the same as used in Korkiakoski et al. (2017). For the fits, the least-squares method was used through the "polyfit" function of NumPy library for the linear regression and the "curve_fit" function of SciPy library for the nonlinear fits.

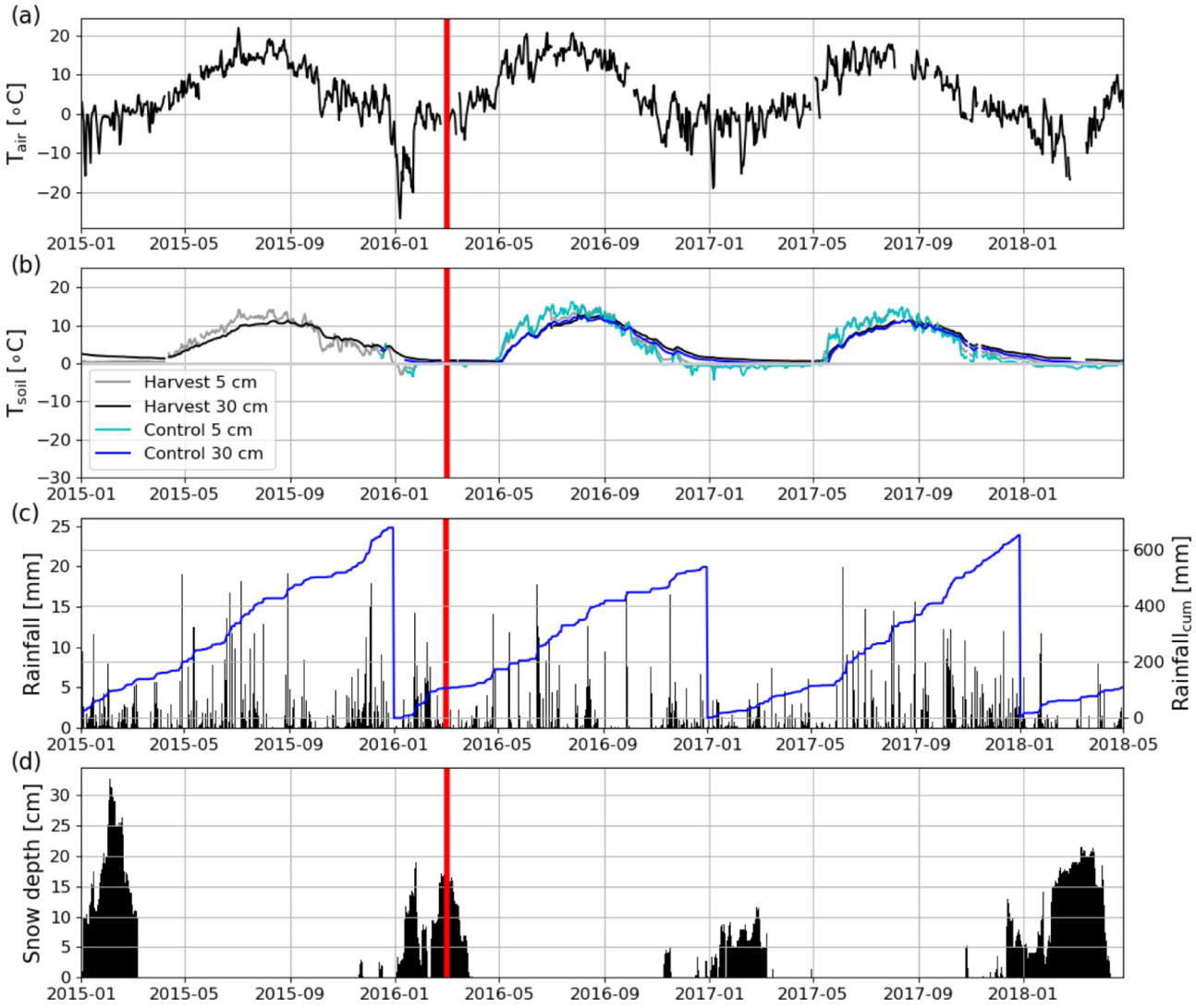

Fig. 2. Time series of (a) daily mean air temperature (T_{air}) and (b) hourly mean soil temperatures at 5 and 30 cm depths (T_{soil}) measured at the harvest and control sites, and (c) daily precipitation sum (black) and annual cumulative rainfall sum (blue), and (d) daily mean snow level recorded at the Jokioinen observatory (35 km northwest of Lettosuo) in January 2015 – April 2018. The harvesting (red vertical line) was carried out in February–March 2016. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

2.6. Statistical analysis

Linear mixed-effect model was used for testing the statistical significance of differences in daily mean CH_4 and N_2O fluxes between the harvest and control sites, and between the years for both automatic and transect chamber measurements. In both cases, the chamber points were treated as a random effect. The linear mixed-effect model was carried out with the R programming language (R Core Team, 2018, version 3.5.0) using 'lme4' package (Bates et al., 2015). The normality of the model residuals was visually checked using quantile-quantile plot (Q-Q plot) method. Tukey's HSD post hoc test was used for comparison of harvest and control sites, and different measurement years for CH_4 and N_2O fluxes.

The linear mixed-effect model was also used for explaining the CH_4 and N_2O fluxes. The logging trail (*LT*, i.e. was the point located on the logging trail or not), dry mass of the needles (DM_n), total dry mass of the logging residues (DM_{tot}, including needles, and branches and twigs of pines, spruces and birches), water table level (*WTL*) and soil temperature at 5 cm (*ST*) were included as fixed effects in the initial model. Chamber points were treated as a random effect (u). The best model was selected by using stepwise selection. We started with a full model and reduced the number of variables one by one using the Akaike information criterion (AIC) as the criteria, which was conducted using the drop1 function in R. The initial model including all factors was:

$$F_{GHG} = \beta_0 + \beta_1 LT + \beta_2 DM_n + \beta_3 DM_{tot} + \beta_4 WTL + \beta_5 ST + \beta_6 (u + e) \quad (1)$$

where F_{GHG} is either CH_4 or N_2O flux, e is the model error, β_0 is the intercept of the model, and parameters from β_1 to β_6 are the regression coefficients of the explaining variables. We used 95% confidence interval ($p < 0.05$) to determine whether the results were statistically significant.

3. Results

3.1. Meteorological and hydrological conditions

The mean annual air temperature at the nearby (Jokioinen, 35 km northwest of Lettosuo) weather station before the harvest in 2015 was 6.2 °C, which was about 1 °C higher than the post-harvest mean temperatures in 2016, 2017 (Table 1; Fig. 2a). The annual temperatures of these three years were higher than the long-term (1981–2010) average at Jokioinen weather station (Table 1), which was also reflected in soil temperatures (Fig. 2b). All three winters (DJF) during the measurement period were warmer than the long-term average (Table 1). The mean summer (JJA) temperatures were variable; summers 2015 and 2017 were 0.8 °C and 1.1 °C cooler, respectively than the long-term average (Table 1), while the mean summer temperature in 2016 was similar to the long-term average. In addition, compared to the precipitation at Jokioinen (Table 1), 2015 and 2017 were slightly wetter while 2016 was drier (Fig. 2c; Table 1). In particular, autumns (SON) 2016 and winter 2016–2017 were much drier, while the springs and summers, were quite similar to the long-term average.

The snow cover was shallower than the long-term average annual maximum every winter during the measurement period (Table 1; Fig. 2d). Only the winter before starting the measurements (i.e. the beginning of 2015) had thicker snow cover than the long-term average annual maximum.

The seasonal inspection of pre-harvest soil temperatures at 5 and 30 cm depth shows that the temperatures were 0.4 °C higher at the control than at the harvest site in summer 2015 (Fig. 2b). However, the soil temperatures were on average 0.3 °C higher at the partially harvest site in both post-harvest summers in 2016 and 2017 as compared to the control site. The summertime soil surface temperatures measured under the logging residues had smaller diel amplitude than the 2 m air temperature measured at the harvest site, but larger than the soil temperatures (Fig. S3). Also, the diel amplitude of the 5 cm soil temperature was smaller when there were residues on top of the soil. The monthly mean summertime diel amplitude of the 2 m air temperature varied within 7.3–11.2 °C, while the surface temperature under the residues varied within 4.0–6.5 °C. The mean differences between the air and surface temperature amplitudes were 3.9 and 4.3 °C in summer 2016 and 2017, respectively. Similarly, the soil temperatures varied within 1.0–2.0 °C with residues on the surface and within 2.4–3.9 °C without residues. The mean differences between the soil temperatures with and without residues were 1.7 and 1.5 °C for summer 2016 and 2017, respectively.

Comparing the summertime PAR measured from six points at each site during daytime (06:00 – 18:00, UTC + 3) revealed that the light conditions between the sites were quite similar (harvest mean: 41 μmol m^{-2} s^{-1}, control mean: 32 μmol m^{-2} s^{-1}). However, after the harvest, the amount of PAR reaching the surface increased markedly (Fig. S4), averaging at 104 and 109 μmol m^{-2} s^{-1} in 2016 and 2017, respectively. At the control site, the PAR remained similar to the pre-harvest mean (2016: 31 μmol m^{-2} s^{-1}, 38 μmol m^{-2} s^{-1}).

The mean WTL in the harvest (–48 cm) and control (–49 cm) transects were similar during the pre-treatment summer (JJA) in 2015 (Fig. 3). The mean WTL at the harvest site was 10 cm and 14 cm higher than at the control site in summer 2016 and 2017, respectively. There was, however, a lot of spatial variation around the site during summers. Outside the transects, the WTL typically varied within –15 and –45 cm, and –25 and –75 cm (Fig. S5) in the harvest and control sites, respectively.

WTL was higher in the middle of the strip (22.5 m from the ditch) at the harvest site, inferred from transect WTL measurements (Fig. S6). On average, the summertime WTL was 23 cm higher in the middle of the strip when compared to the point located 4 m from the ditch. However, at the control site, there was not such a clear trend, and generally, the WTL was higher at the 12 m point while the WTL was similar at the rest of the points.

3.2. Automatic chamber fluxes

3.2.1. CH_4 exchange

CH_4 fluxes showed an apparent seasonal variation at both harvest and control sites (Fig. 4a). Generally, the fluxes were close to zero in the February–April period, but noticeable sink started to develop in May after the soil surface had thawed. The soil sink continued increasing until September when the sink started to decrease with decreasing soil temperatures. In February and March, the CH_4 flux was close to zero. CH_4 emissions were rare and observed only after some heavy rainfall events.

Here we report the CH_4 balances in 6-month periods because this allowed us to compare the balances after the harvest directly to those before the harvest. Before the harvest in 2015, the mean half-year (July–December) CH_4 balances were –156 ± 10 mg CH_4 m^{-2} (± uncertainty, see Sect 2.6) and –167 ± 8 mg CH_4 m^{-2} 6 $months^{-1}$ for harvest and control site (Table S5), respectively; and did not differ significantly from each other. The variation between the measurement points was high at both the harvest and the control sites (from –60 ± 5 to –248 ± 57 mg CH_4 m^{-2} and from –75 ± 10 to –215 ± 36 mg CH_4 m^{-2}), respectively. After the harvest, in 2016, the half-year net CH_4 uptake decreased on average by 44% at the harvest site and by 31% at the control site (Table S5). The net uptake decreased in all 12 measurement points, but the relative and absolute changes varied. Even though the decrease in net CH_4 uptake from 2015 was significant ($p < 0.001$) in both sites, the harvest and control sites still did not differ significantly from each other in 2016. In 2017, the net CH_4 uptake at the harvest site was similar to the 2016 values in three measurement points while in the three other points the net uptake decreased further (Table S5). In contrast to the harvest site, the net CH_4 uptake at the control site in 2017 was in between the uptake rates measured in 2015

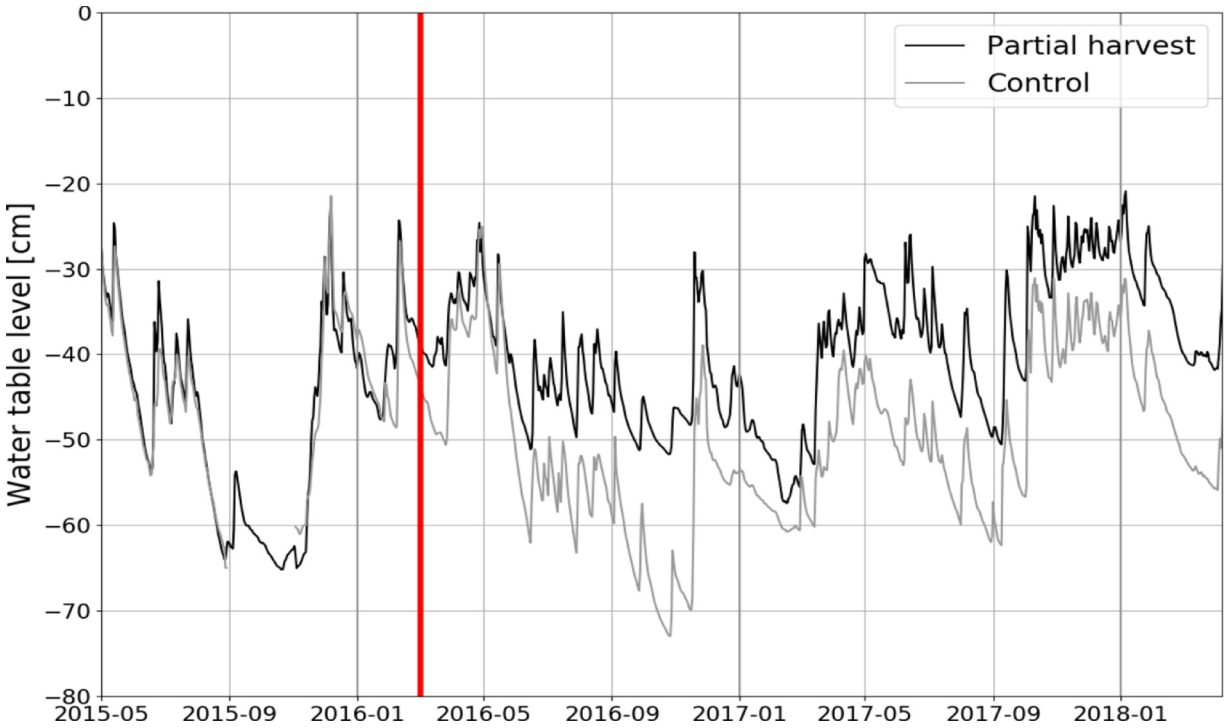

Fig. 3. Daily mean water table level in the control (grey) and harvest site (black line) averaged over all the forest transect measurement points (4, 8, 12 and 22.5 m from the ditch) from May 2015 to April 2018. The harvesting (red vertical line) was carried out in February-March 2016. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

and 2016 (Table S5). However, the difference in net CH_4 uptake was still not statistically significant between the sites on half-annual nor seasonal timescales; in other words, the harvest did not have a significant impact on the soil CH_4 uptake, when inferred from the automatic chamber measurements.

3.2.2. N_2O exchange

At the harvest site, the magnitude of the N_2O fluxes varied markedly between the measurement points. However, the temporal dynamics remained quite similar before and after harvesting. The emissions were typically highest in summer (JJA, Fig. 4b) when on average 34% (April 2016 – March 2017) and 47% (April 2017 – March 2018) of the annual emissions were recorded. In autumn (SON), the emissions were relatively low, but typically in December or January, the emissions increased again after the soil froze ($T_{5cm} < 0$ °C). These elevated emissions after soil freezing could last up to three months and have a large contribution to the annual balance. The contribution to annual balance varied between 15 – 48% when inspecting the annual period of April

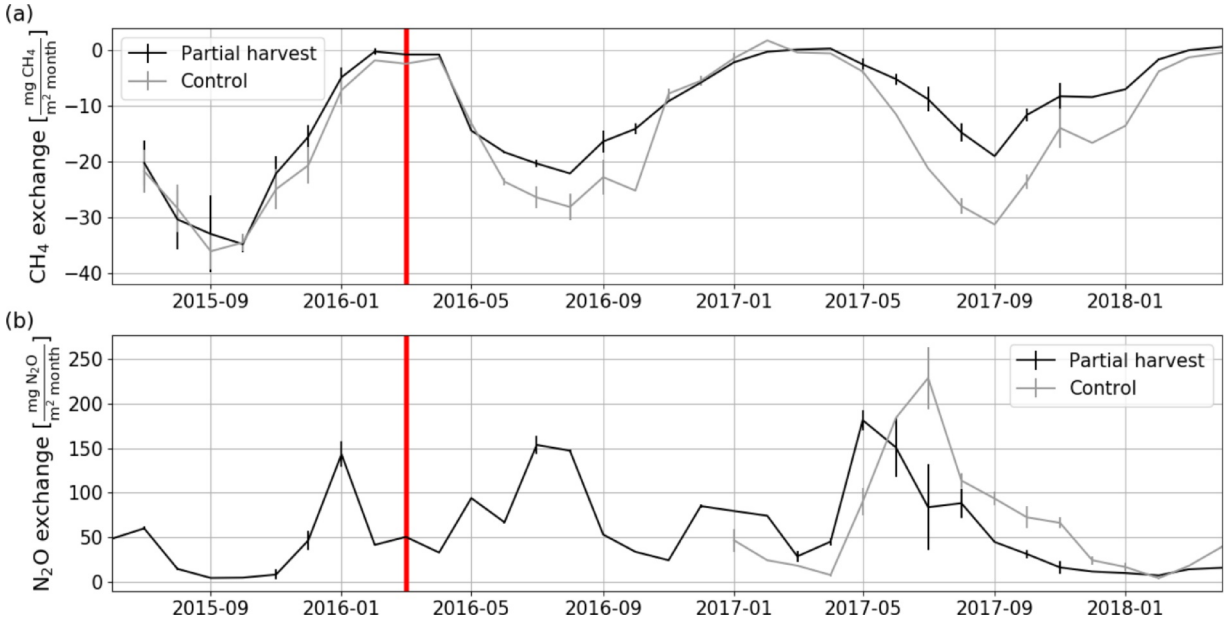

Fig. 4. Monthly mean sum time series of CH_4 (a) and N_2O (b) exchange of six automatic chambers at the harvest (black) and control (grey) sites in June 2015 – April 2018. The error bars show the estimated uncertainty of the monthly balance (Appendix C). The harvesting (red vertical line) was carried out in February-March 2016. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

2016 – March 2017. The contribution of these elevated winter emissions was higher in points with relatively low annual emissions. Unlike in previous years, no elevated emissions in winter were observed in late 2017 or early 2018. Between elevated emissions in winter and summer, there was a period of low emissions typically in March and April similarly to autumn.

Before the harvest, the half-year (July–December) N_2O balance at the harvest site averaged at 138 ± 12 mg N_2O m^{-2} (± uncertainty, see Appendix C), with considerable variation between the measurement points (52 ± 17 – 222 ± 23 mg N_2O m^{-2}). After harvesting in 2016, all six points recorded increased emissions, but the relative and absolute increases varied (Table S5). The mean post-harvest N_2O emission across all the points in 2016 was 498 ± 10 mg N_2O m^{-2}, and the change from 2015 was statistically significant ($p < 0.001$). The mean N_2O balance at the harvest site in 2017 was in between the balances of 2015 and 2016 at 277 ± 52 mg N_2O m^{-2} and was significantly different from previous years ($p < 0.001$).

At the control site, we only had automatic chamber measurements in 2017. The half-year cumulative emissions from the measurement points C1-C6 averaged at 600 ± 39 mg N_2O m^{-2} varying from 145 ± 20 to 1449 ± 178 mg N_2O m^{-2} (Table S5). Comparing this same period to the measurements at the harvest site (mean: 277 ± 52 mg N_2O m^{-2}) showed that the emissions from the control site were not significantly different during the latter half of 2017. Also, comparing annual periods of 1/2017 – 12/2017 and 4/2017 – 3/2018 did not reveal significant differences between the sites.

3.3. Manual chamber fluxes at the transects

Both the harvest and control site acted as CH_4 sinks both before and after the harvest (Fig. 5 and S7a). CH_4 emissions were observed at 12 out of 16 possible points across the sites at least once, but only at five of them emissions were observed more than once. Again, there was a lot of spatial and temporal variation in fluxes, which varied between –484 and 24 μg CH_4 m^{-2} h^{-1}, and –158 and 51 μg CH_4 m^{-2} h^{-1} during summer (JJA) at the harvest and control site, respectively. At the harvest site, measurement points 4 m from the ditch had significantly higher sink ($p < 0.05$) in all calendar years than the points farther from the ditch. In contrast, at the control site, the fluxes generally did not differ significantly between the measurement points. Even though the mean pre-harvest summertime CH_4 uptake rate at the control site was only 49% of that of the harvest site (harvest: –95 ± 34 μg CH_4 m^{-2} h^{-1}, control: –47 ± 11 μg CH_4 m^{-2} h^{-1}; ± standard error of the mean; Fig. 5), the sites were not significantly different. The net uptake decreased after the harvest by 41% (mean flux: –56 ± 6 μg CH_4 m^{-2} h^{-1}) in 2016 and remained so also in 2017, both post-harvest summers were significantly different from the pre-harvest summer (2016: $p < 0.01$, 2017: $p < 0.001$). At the control site, the post-harvest CH_4 sink was similar to the pre-harvest sink. Comparing the harvest and control sites against each other showed that they did not differ significantly in either of the post-harvest summers.

The pre- and post-harvest N_2O fluxes were spatially and temporally highly variable at both harvest and control site (Fig. S7b). Before the harvest, the mean summertime fluxes were 178 ± 109 μg N_2O m^{-2} h^{-1} (± standard error of the mean) and 38 ± 33 μg N_2O m^{-2} h^{-1} at the harvest and control site, respectively. However, the two sites were not significantly different with respect to mean summer (JJA) N_2O fluxes before the harvest. At the harvest site, the fluxes did not change significantly after the harvest. At the control site, summertime fluxes in 2017 (mean: 251 ± 37 μg N_2O m^{-2} h^{-1}) were significantly higher when compared to two previous summers (2015: $p < 0.01$, 2016: $p < 0.001$). However, the post-harvest summertime fluxes in 2016 and 2017 did not differ significantly between the harvest and control sites. In conclusion, the transect measurements did not show any significant harvest-related changes in N_2O fluxes.

3.4. Logging trail and logging residue fluxes

Three of the four measurement point types acted most of the time as CH_4 sink; fluxes were varying mostly between –40 and 0 μg CH_4 m^{-2}

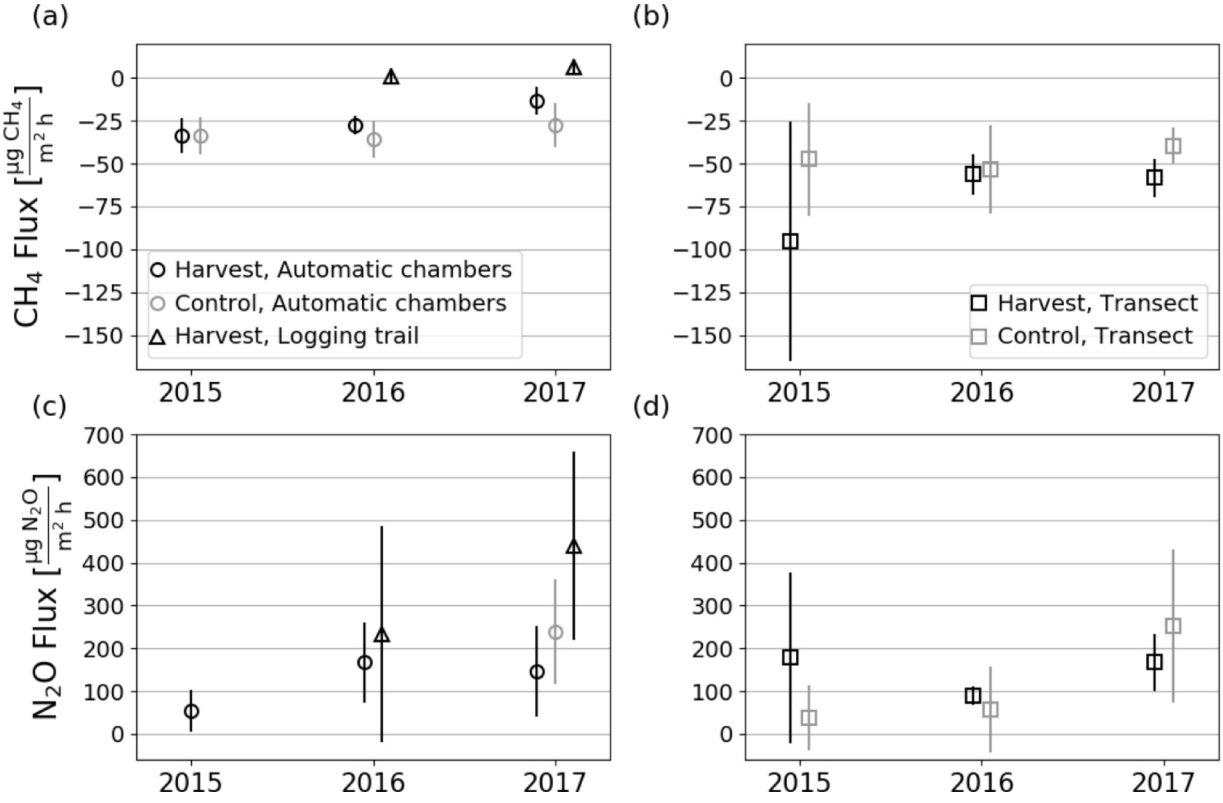

Fig. 5. Summertime (JJA) mean of the hourly CH_4 (a, b) and N_2O (c, d) fluxes (± standard deviation) measured at the harvest (black) and control site (grey) by automatic chambers (circle), manual chambers at the transect (square) and the logging trail (triangle).

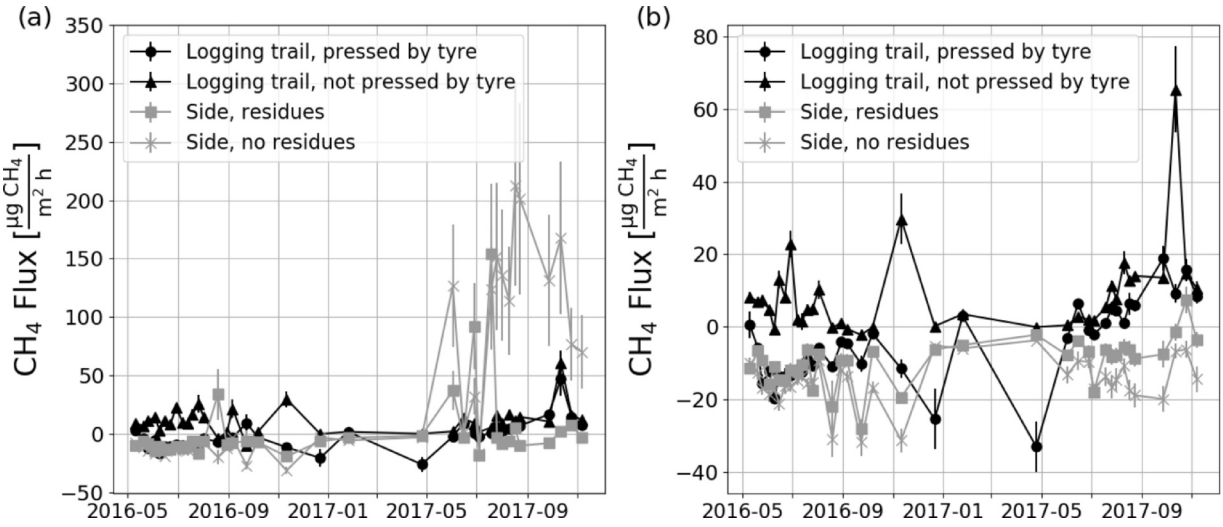

Fig. 6. Daily mean (a: $n = 5$, b: $n = 4$) CH_4 flux (± standard error of the mean) time series measured with manual chambers on the tyre pressed (black circle) and not tyre pressed (black triangle) part of the logging trail, and on the side of the logging trail with small amount (grey square) and without (grey cross) logging residues in May 2016 – November 2017. The time series represent data with (a) and without (b) plot #5.

h^{-1}, while the points located in the uncompressed part of the logging trail commonly acted as CH_4 sources (Fig. 6). In 2016, small CH_4 emissions (< 30 μg CH_4 m^{-2} h^{-1}) were sporadically observed also from the compressed points on the logging trail, whereas in 2017 the emissions from the compressed points were frequent from all plots except #2 (Fig. S8). The fluxes from the logging trail were also significantly higher when compared to the transect measurements at the harvest site ($p < 0.001$; Sect. 3.3). The points on the side of the logging trail were mostly CH_4 sinks in 2016 in all but plot #5 where several relatively high CH_4 emissions were recorded (> 600 μg CH_4 m^{-2} h^{-1}; Fig. S8). In 2017, small emissions were also observed on plots #3 and #4 from the points on the side of the logging trail. Also, the emissions on plot #5 increased from the previous year (Fig. 6a; Fig. S8). When inspecting the whole dataset, no significant differences between the point types were found. Also, none of the inspected variables influenced the CH_4 fluxes significantly. Because plot #5 was noticeably wetter than the other plots, the analysis concerning the dependence of CH_4 fluxes on environmental variables was also carried out without plot #5 (Fig. 6b). Removing plot #5 enhanced the statistical significance observed in CH_4 flux between the points in and outside the logging trail ($p < 0.01$; Table S6). Also, now using the whole dataset the points on the uncompressed part of the logging trail differed significantly from both points outside the trail. Without plot #5, explanatory factors for CH_4 flux were found, and the best model indicated that the CH_4 flux was most influenced by WTL ($p < 0.001$) and the location of the measurement point in or outside of the logging trail ($p < 0.04$). The model explained 36% of the variation in CH_4 fluxes, and the contribution of the fixed effects was 24%.

All the plot types acted as N_2O source (Fig. 7) during the whole measurement period. During the first two months of the measurements (May–June 2016) the emissions within plots were somewhat similar, but N_2O emissions started to increase in July commonly from the points located on the logging trail (Fig. S9). Again, the emissions were highly variable varying within two orders of magnitude, and none of the plot types differed significantly from each other when inspecting the whole dataset. However, the fluxes from the points on the logging trail were significantly ($p < 0.03$) higher than those outside of the trail. Generally, the daily mean N_2O flux over all the points outside the trail was < 200 μg N_2O m^{-2} h^{-1} while the mean emissions from the trail were more variable (generally between: 100 – 900 μg N_2O m^{-2} h^{-1}; Figs. 7 and S9). Also, the logging trail points had significantly higher emissions than the emissions measured from the forest floor with automatic chambers at harvest ($p < 0.01$; Sect. 3.2.2; Fig. 5). The best model

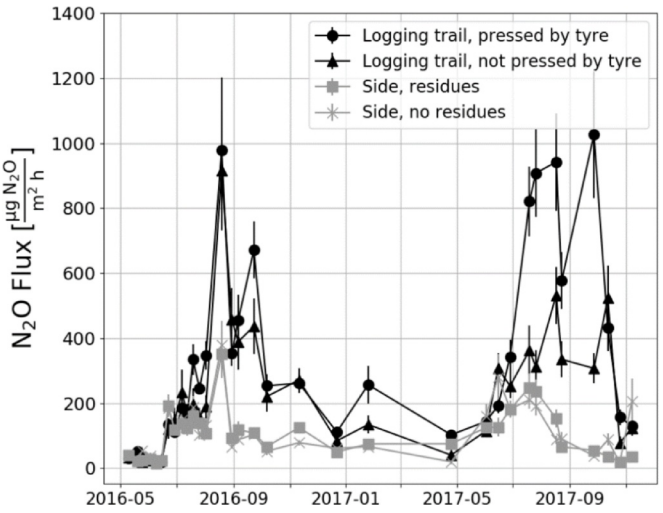

Fig. 7. Daily mean ($n = 5$) N_2O time series measured with manual chambers on the tyre pressed (black circle) and not tyre pressed (black triangle) part of the logging trail, and on the side of the logging trail with small amount (grey square) and without (grey cross) logging residues in May 2016 – November 2017. The error bars show the standard error of the mean.

explaining the N_2O fluxes showed that the N_2O flux was most influenced by WTL ($p < 0.001$) and 5 cm soil temperature ($p < 0.01$) followed by the location of the point on the logging trail ($p < 0.02$) and the total dry mass of the residues ($p < 0.03$). The model explained 48% of the variation in N_2O fluxes, but it was mostly due to random effects as the r^2 of the fixed effects was only 14% (Table S6).

3.5. Ditch fluxes

The mean CH_4 emissions from the ditches were significantly ($p < 0.001$) higher during the whole measurement period at the harvest site than at the control site. However, as the ditches were so different (water-filled at the harvest and moss-filled at the control site) and the number of measurement points in the ditches was small, it was not sensible to estimate the effect of harvest on ditch fluxes. The means of all the measured CH_4 fluxes from the ditches were 2754 ± 642 μg CH_4 m^{-2} h^{-1} (± standard error of the mean) and 127 ± 45 μg CH_4 m^{-2} h^{-1}, and the fluxes varied within 0–10894 μg CH_4 m^{-2} h^{-1} and

–238–1905 μg CH_4 m^{-2} h^{-1} (Fig. S10a) at the water- and moss-filled ditch, respectively. The fluxes at the water-filled ditch were more temporally variable than at the moss-filled ditch. Also, the fluxes were mostly zero in both ditches during November–March. In the moss-filled ditch, occasional net CH_4 uptake was measured, but fluxes were always zero or above at the water-filled ditch. No statistically significant differences were found when comparing the CH_4 fluxes between the years or summers in the water-filled ditch. However, the year and summer 2015 were significantly ($p < 0.05$) different from both respective periods in 2016 and 2017 at the moss-filled ditch.

The N_2O fluxes from the ditches were low; the mean fluxes over the whole measurement period were 13 ± 5 μg N_2O m^{-2} h^{-1} (± standard error of the mean) and 40 ± 19 μg N_2O m^{-2} h^{-1} (Fig. S10b) at the water- and moss-filled ditch, respectively. The fluxes were more variable in the moss-filled ditch ranging from –140 μg N_2O m^{-2} h^{-1} to 690 μg N_2O m^{-2} h^{-1} than in the water-filled ditch where the fluxes varied within –39 and 150 μg N_2O m^{-2} h^{-1} (Fig. S10b). The N_2O fluxes in the moss-filled ditch were not significantly different between years. However, in the water-filled ditch, the year 2017 was significantly different from the respective periods. In the water-filled ditch, the fluxes were < 54 μg N_2O m^{-2} h^{-1} in 2015 and 2016, but in 2017 several emission peaks up to 150 μg N_2O m^{-2} h^{-1} were observed. In addition, summer 2015 was the only period when water- and moss-filled ditches differed significantly ($p < 0.01$) in N_2O fluxes.

3.6. Upscaling CH_4 and N_2O fluxes to ecosystem level

Upscaling the CH_4 fluxes on ecosystem-scale using all the chamber measurements in previous chapters (Sects. 3.2, 3.3, 3.4), except data from logging trail plot #5, revealed that both the harvest and control sites were around CH_4 neutral during summertime (Table 2). In 2016, the harvest site was a small CH_4 source (10 ± 21 mg CH_4 m^{-2}) as the emissions from the ditches (43 ± 21 mg CH_4 m^{-2}) were larger than the net CH_4 uptake in the forest floor (–31 ± 2 mg CH_4 m^{-2}). During the following summer, the emissions from the ditches increased, and the forest floor uptake decreased; therefore, the CH_4 balance of the harvest site increased to 47 ± 33 mg CH_4 m^{-2}. The fluxes from the logging trails had no significant impact on ecosystem-level balance during the summers of 2016 or 2017. The estimated forest floor uptake was more substantial at the control site than at the harvest site, which resulted in smaller net CH_4 emissions at the control site during both summers. In 2016, the control site acted as a net CH_4 sink (–18 ± 22 mg CH_4 m^{-2}) while in 2017, the site was a small source (19 ± 33 mg CH_4 m^{-2}).

Both the harvest and control sites acted as N_2O sources during the summers (Table 2). At the harvest site, forest floor contributed the most to the ecosystem N_2O balance (2016: 65%, 2017: 62%), while the rest of the ecosystem-level N_2O emissions originated from the logging trails. The fluxes from the ditches were insignificant. At the control site, only the emissions from the undisturbed forest floor were significant on ecosystem-scale. The N_2O emissions at the harvest site (368 ± 29 mg N_2O m^{-2}) were larger than the emissions at the control site (120 ± 39 mg N_2O m^{-2}) in 2016. However, in 2017 the emissions increased in both sites, but the increase at the control site was so large that the total emissions became similar between both sites (harvest: 533 ± 114 mg N_2O m^{-2}, control 495 ± 46 mg N_2O m^{-2}). When considering the climatic effect of these gases in terms of their global warming potential up to 100 years with climate-carbon feedbacks (CH_4: GWP_{100} = 34; N_2O: GWP_{100} = 298; IPCC, 2013), the total emissions of CH_4 were on average only 0.8% of the N_2O emissions (Table 2).

4. Discussion

4.1. Partial harvest has lower N_2O and CH_4 emissions than clearcutting

In this study, we found that the partial harvesting did not have a significant impact on CH_4 and N_2O fluxes in a drained nutrient-rich peatland forest when comparing the pre- and post-harvest situations at the harvest site. Decrease in CH_4 uptake have previously been observed in partially harvested boreal upland forests (Sundqvist et al., 2014), but increase in CH_4 uptake was recorded in a Mediterranean upland forest (Mazza et al., 2019). Papers studying the effects of thinning on N_2O fluxes are scarcer, but increases in emissions have been observed (Li et al., 2010). However, it is difficult to compare different thinning studies as the method of thinning varies, which affects the measured greenhouse gas fluxes (Mazza et al., 2019). According to the previous study at the site made in the same time period as this study (Korkiakoski et al., 2019), clearcutting increased N_2O emissions markedly and turned the clear-cut forest floor from CH_4 sink into a small CH_4 source, and raised WTL by 23 cm. Despite a slight increase of 10 – 14 cm in water level, the forest floor at the partial harvest site remained as a small CH_4 sink (Fig. 5), which was due to WTL staying mostly under –30 cm (Fig. 3) that has been considered a limit above which CH_4 emissions may occur in peatland forests (Ojanen et al., 2010, 2013). Also, the mean annual CH_4 balance measured by the automatic chambers (from –87 ± 4 to –169 ± 4 mg CH_4 m^{-2} $year^{-1}$) was smaller than measured at the same forest in 2011–2013 (–219 mg CH_4 m^{-2} $year^{-1}$; Korkiakoski et al., 2017), but still within typical uptake rates of 10–970 mg CH_4 m^{-2} $year^{-1}$ reported in drained peatland forests in Finland (Lohila et al., 2011; Minkkinen et al., 2007; Ojanen et al., 2010). It should be noted, however, that the CH_4 emissions (mean balance: 0.8 g CO_2-eq m^{-2} $summer^{-1}$) at Lettosuo after partial harvesting and even after clearcutting (Korkiakoski et al., 2019) were negligible compared to N_2O emissions (mean balance: 113 g CO_2-eq m^{-2} $summer^{-1}$) when considering their climatic impact in terms of the

Table 2
Upscaled CH_4 and N_2O balances (± uncertainty, see Sect. 2.5) from different surface types weighted by their respective share of surface area for summers 2016 and 2017. The balances are also reported in CO_2-equivalents in terms of the global warming potential up to 100 years.

CH_4 flux [mg CH_4 m^{-2} $summer^{-1}$]					
Area & Year	Total	Ditch	Forest	Logging trail	Total [g CO_2-eq m-2 summer-1]
Harvest 2016	10 ± 28	43 ± 28	-31 ± 2	–1.8 ± 0.9	0.3 ± 1.0
Harvest 2017	47 ± 43	64 ± 43	-19 ± 4	2.1 ± 0.5	1.6 ± 1.5
Control 2016	–18 ± 28	43 ± 28	-61 ± 5	NA	0.6 ± 1.0
Control 2017	19 ± 43	64 ± 43	-45 ± 3	NA	0.6 ± 1.5
N_2O flux [mg N_2O m^{-2} $summer^{-1}$]					
Area & Year	Total	Ditch	Forest	Logging trail	Total [g CO_2-eq m-2 summer-1]
Harvest 2016	368 ± 29	-0.2 ± 0.3	240 ± 17	129 ± 24	110 ± 9
Harvest 2017	533 ± 114	0.6 ± 0.4	331 ± 111	201 ± 26	159 ± 34
Control 2016	120 ± 39	-0.2 ± 0.2	120 ± 39	NA	36 ± 12
Control 2017	495 ± 46	0.6 ± 0.3	494 ± 46	NA	148 ± 14

global warming potential up to 100 years.

Based on both manual and automatic chamber data, the effect of partial harvest on N_2O fluxes was not significant. Nevertheless, this means that the partial harvest did not cause a similar increase in N_2O emissions as the clearcutting of the same forest (Korkiakoski et al., 2019) where the change was apparent. Smaller changes in N_2O fluxes were expected as clearcutting removes more trees, raises the WTL, produces more logging residues and severely disturbs ground vegetation, which releases a large amount of reactive nitrogen (NO_3^-, NH_4^+). Thus, an increased amount of mineral N is converted to N_2O as the uptake by vegetation is diminished. Partial harvesting produces less logging residues as not all the trees are harvested, and ground vegetation is less disturbed, except on the logging trails where most of the vegetation is covered by logging residues. Indeed, N_2O emissions from the logging trails were significantly larger than the emissions outside the logging trails (Fig. 7). Comparing these two harvesting methods at the same site shows that the mean N_2O flux measured by the automatic (368 ± 10 μg N_2O m^{-2} h^{-1}) and manual chambers (178 ± 109 μg N_2O m^{-2} h^{-1}) after harvesting in summer 2016 is noticeably smaller than at the clear-cut site where the fluxes increased markedly from about zero up to 1000 μg N_2O m^{-2} h^{-1} on average (Korkiakoski et al., 2019). Comparing the automatic chamber measurements made in this study shows that the mean annual N_2O balances at Lettosuo (0.69 ± 0.06 – 0.92 ± 0.04 g N_2O m^{-2} $year^{-1}$) are at the higher end of the range (–0.03–0.92 g N_2O m^{-2} $year^{-1}$) previously reported at drained peatland forests in Finland (Lohila et al., 2011; Ojanen et al., 2010).

4.2. The effect of partial harvesting on CH_4 fluxes

Partial harvesting did not significantly affect the forest floor CH_4 fluxes (Sect. 3.2.1; 3.3). There were significant differences between the years, but these were unlikely due to the harvest as similar changes were observed at the control site as well. Harvesting raised the WTL on average by 10–14 cm due to decreased transpiration. Higher WTL makes the oxic layer on the top soil thinner and conditions for CH_4 oxidation less favourable, which allows less CH_4 produced in the anoxic layer to be oxidised before reaching the atmosphere (e.g. Lai, 2009). The decreases in net half-year CH_4 uptakes in both harvest (2016: 49%) and control (2016: 26%) sites (Sect. 3.2.1) are likely attributed to higher WTLs during autumn 2016 than in 2015, which was further increased at the harvest site due to reduced transpiration after the removal of transpiring trees. Removal of the trees also increased photosynthetically active radiation under the canopy, which could have dried the exposed soil surface due to increased soil temperatures during the daytime (Londo et al., 1999). This could have counteracted parts of the decreased CH_4 oxidation due to WTL rise after the harvest. Soil temperature is also known to affect CH_4 production and oxidation (e.g. Malyan et al., 2016), but the difference in temperature between the sites was rather small (2015: < 0.4 °C; 2016 and 2017: < 0.3 °C). A previous study concerning the changes in forest floor GHG fluxes after clearcutting at Lettosuo concluded that the site turned from CH_4 sink into a small CH_4 source due to a 23 cm rise of mean summertime WTL to the level of –24 cm (Korkiakoski et al., 2019). Partial harvesting raised WTL less (10–14 cm) than the clearcutting, keeping WTL generally below –30 cm, which did not significantly affect the soil CH_4 sink. Similar results have been found from a boreal forest in central Sweden where clearcutting turned the site to a CH_4 source, but after the thinning, the site continued acting as a CH_4 sink (Sundqvist et al., 2014).

The net CH_4 emissions were higher from the plots located on the logging trail than at the plots outside of the logging trail (Fig. 6b). The emissions could have increased due to the compression of soil by heavy machinery used in the tree harvesting. The compression of soil lowered the soil surface, probably decreasing the thickness of the oxic layer and the amount of oxygen in the soil, which then either reduced CH_4 oxidation or increased production by creating anoxic spots in the soil. Contrary to other plots, the CH_4 emissions at plot #5 were higher from the plots outside of the logging trail. This was probably caused by higher soil moisture leading to the growth of *Eriophorum vaginatum* which acts as a conduit for CH_4 release (Greenup et al., 2000) and is a relatively high CH_4 source compared to other vegetation communities (Minkkinen and Laine, 2006). Logging residues change the soil environmental conditions by decreasing soil temperature and conserving soil moisture (Ojanen et al., 2017; Roberts et al., 2005), which both are known to affect CH_4 fluxes. We also observed a dampened diel cycle on the soil surface under the residues when compared to the air temperature. However, we did not measure soil moisture close to these measurement points, and we did not find a significant relationship between the soil temperature nor any other environmental variables and CH_4 fluxes. However, after ignoring the wet plot #5, CH_4 flux was found to be significantly influenced by the location on the logging trail and by the WTL, which affects the thickness of the aerated layer in the soil. Our findings agree with Mäkiranta et al. (2012) that logging residues do not impact significantly to CH_4 fluxes. At our site, the changes in soil physical properties on the logging trail were behind the higher CH_4 emissions at the logging trail than in the forest.

Even though the CH_4 fluxes were small, there was some spatial variation observed on different surface types (Sect. 3.6). Therefore, we combined all our measurements to upscale the measured fluxes to ecosystem level by using WTL as a proxy. As a result, the cumulative CH_4 balances at the partial harvest forest floor during summers 2016 and 2017 were –31 ± 2 mg CH_4 m^{-2} and –19 ± 4 mg CH_4 m^{-2}, respectively, which were 49% and 34% lower than the sinks estimated from the automatic chamber measurements. This was due to higher, but also variable WTL inside the harvested site. The contribution of logging trails to the ecosystem balance was negligible due to a combination of small fluxes and relatively small surface area (20% of total area). However, even though the ditches constitute only 2.4% of the total harvested area, they could play a considerable role in the total ecosystem CH_4 balance due to their relatively high CH_4 emissions. The CH_4 emissions from the ditches on the ecosystem level were higher than the forest floor sink at the harvest site during both summers and also at the control site in summer 2017 (Table 2). However, the ecosystem-level ditch fluxes had high uncertainty due to high variability in fluxes, and the fact that only two ditches were measured.

In conclusion, the partial harvest site was likely a CH_4 neutral or a small source due to high emissions from the ditches and decreased sink in the forest floor. However, it should be noted that even though the uncertainties were relatively large, even in the worst-case scenario, the site would still be only a small source of CH_4. Also, the climatic impact of CH_4 emissions in terms of global warming potential over 100 years (mean: 0.8 ± 0.6 g CO_2-eq m^{-2} $summer^{-1}$) is negligible when compared against N_2O emissions (mean: 113 ± 10 g CO_2-eq m^{-2} $summer^{-1}$) at Lettosuo.

4.3. The effect of partial harvesting on N_2O fluxes

Forest floor N_2O fluxes were temporally and spatially variable at both harvest and control sites, and no significant difference between them was found due to the harvest (Sect. 3.2.2; 3.3). Based on the data from the automatic chambers at the harvest site, the fluxes were significantly different between all the years. However, no such differences were found in the manual chamber measurements. The contradiction between the automatic chamber and manual chamber measurements was at least partly due to the high spatial variation between the plots and due to lower measurement interval in the transect. Despite the slight, unexplained discrepancy between the methods, it is good to note that at the clear-cut site of the same forest (Korkiakoski et al., 2019), a large increase in N_2O fluxes was observed during the same time period using the same manual measurement system as in this study. Comparing the results of this study to Korkiakoski et al. (2019) suggests that the effect of the partial harvest on N_2O fluxes was negligible as it was not observable by manual nor automatic measurements.

An interesting result from the automatic chambers was the bimodal behaviour of N_2O fluxes during the year, showing high emissions in summer and in winter (Fig. 4b). This phenomenon has been observed in multiple studies in upland and agricultural sites, and the emissions during these cycles can be a substantial part of annual N_2O balance (Luo et al., 2012; Papen and Butterbach-Bahl, 1999; Peng et al., 2019; Röver et al., 1998). Large emissions after soil thawing can be explained by N_2O accumulation under the ice (Peng et al., 2019; Teepe et al., 2001). Also, increased emissions have been observed when the soil is frozen, which could occur due to increased amount of substrates during soil freezing (Herrmann and Witter, 2002; Matzner and Borken, 2008; Song et al., 2017). These substrates could be used by microbes living in small unfrozen water films in soil pores (Congreves et al., 2018; Papen and Butterbach-Bahl, 1999).The manual chamber measurements showed significant differences in N_2O fluxes between the summers at the control site, but not at the harvest site. This might be attributed to the counteracting effect of harvesting and the environmental conditions: summer 2016 was drier than 2015 which inhibited N_2O emissions at the control, however at the harvest site, the water-level drawdown due to dry conditions was not marked enough to inhibit emissions. The drier summer might thus explain the smaller N_2O emissions in 2016 at the control because low soil moisture has been attributed to relatively low N_2O emissions (Ciarlo et al., 2007; Dalal et al., 2003; Gelfand et al., 2015).

N_2O emissions were higher on the logging trail than on the side of the logging trail (Fig. 7) and on the forest floor measured by the automatic chambers at harvest site (Fig. 5). The N_2O fluxes were significantly correlated with the location on the logging trail, WTL, soil temperature and the mass of the logging residues. The mass of needles did not significantly correlate with N_2O fluxes. However, the model explained only a small part of the variation in fluxes. Thus, other factors such as soil moisture, which is known to drive N_2O fluxes (e.g. Brown et al., 2012), might have had some effect. Soil moisture could have increased due to soil compression, which enhanced denitrification and N_2O emissions. Also, logging residues stabilised soil temperature and possibly moisture by preventing direct sunlight and evaporation which favour mineralisation and nitrification processes in the organic layer of soil (Roberts et al., 2005; Rosén and Lundmark-Thelin, 1987). Upscaling fluxes to the ecosystem-level revealed that the N_2O emissions from the logging trails were 35–38% of total site balance (Table 2) making the logging trails an essential part of the ecosystem N_2O balance. However, there was an exception with logging trail plot #5 where WTL was around –20 cm or higher. The N_2O emissions at that plot were low because the conditions were too wet for nitrification and N_2O production to occur.

5. Conclusions

Partial harvesting a nutrient-rich peatland forest did not affect CH_4 or N_2O balances considerably, and the site remained as CH_4 neutral and a small source of N_2O even after the harvest. The logging trails acted as a small CH_4 source due to soil compression which likely increased soil moisture and decreased CH_4 oxidation, but on ecosystem-level, logging trail CH_4 fluxes were insignificant. Given the small number of measurements and the large uncertainties related to the ditch CH_4 fluxes, the exact quantification of ditch fluxes was not made. However, CH_4 fluxes in peatland forest seem to have a negligible climatic impact, even after the harvest.

The emissions of N_2O from the logging trails were significantly higher than from the forest surface where no changes were observed after harvesting. The higher emissions from the logging trails could have been caused by soil compression, which increases soil moisture and enhances denitrification. Also, decomposing logging residues release reactive nitrogen and prevent direct sunlight and evaporation, which favour nitrification processes in the organic layer of soil. On ecosystem-level, the emissions from the logging trails were an essential part of the N_2O balance of the site constituting 35–38% of the total emissions while the forest floor emitted the rest. The fluxes from the ditches were insignificant. Despite capturing the high temporal and spatial variability in N_2O fluxes by different chamber measurement methods, and recording increased emissions from the logging trails, no considerable harvesting effect in N_2O fluxes was observed when compared to the control site.

Overall, the results in this study show that partial harvesting of peatland forests causes less CH_4 and N_2O emissions compared to clearcutting. However, the results here show only the impact of partial harvesting for the first two post-harvest years and more measurements are required to make conclusions of long-term effects of partial harvesting in peatland forests.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

This study was funded by the Maj and Tor Nessling Foundation (grant no. 201700450), the Academy of Finland project “Role of upland forest soils in regional methane balance: from catchment to global scales” (project no. 308511), and the Ministry of Transport and Communications through the Integrated Carbon Observing System (ICOS) research.

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.agrformet.2020.108168.

Appendix A. Flux calculation

The fluxes were calculated the same way as described in Korkiakoski et al. (2017) for both gases. Dilution and spectral corrected CH_4 and N_2O concentrations reported by Picarro G1130 and LGR N2O/CO-23d were used to calculate the fluxes. Even though the lag caused by tubing was taken into account, 30 s from the beginning of each closure data was removed (Korkiakoski et al., 2017) because the flow rate varied slightly in time, and it takes time for the air inside the chamber to mix properly. In short, both linear and exponential regression models were first fitted to the mixing ratio time series using the least-squares approach.

After fitting, the mass flux (F) was calculated as

$$F = \left(\frac{dC(t)}{dt}\right)_{t=0} \frac{MPV}{RTA}, \quad \text{(A1)}$$

where $(\frac{dC(t)}{dt})_{t=0}$ is the concentration change over time from a linear or exponential model at the beginning of the closure, M is the molecular mass of CH_4 or N_2O (16.04 and 44.01 g mol^{-1}, respectively), P is air pressure, R is the universal gas constant (8.314 J mol^{-1} K^{-1}), T is the mean chamber headspace temperature during the closure, and V is the air volume of the chamber and the possible collar, and A is the base area of the chamber or collar. The collar height, snow depth and the height of mosses and other vegetation in the chamber headspace volume were taken into account, ignoring the pore space in the soil and snow. Finally, analyser-specific flux limits were determined to choose between the linear and exponential models (Korkiakoski et al., 2017). If the flux calculated with the linear model was smaller than the limit, then this estimate was considered more robust and used in the later analysis. These limits for the automatic chamber system at the harvest site were 2.5 μg CH_4 m^{-2} h^{-1} and 3 μg N_2O m^{-2} h^{-1}. At the control chamber system, the CH_4 limit was the same, but N_2O limit was 5 μg N_2O m^{-2} h^{-1} for the LGR analyser.

At the control site, the LGR and Gasmet instruments were simultaneously connected to the chamber system in July – December 2017 to compare the instruments. The comparison revealed that the 6 min closure time used in the automatic chambers was not long enough for Gasmet to determine the flux by using exponential regression. Therefore, only linear regression was used to calculate the fluxes when using only the Gasmet analyser in the first half of 2017 (Fig. S1). The N_2O fluxes calculated from the Gasmet data were generally underestimated by 10–20%, depending on the flux magnitude when compared to the fluxes estimated from the LGR data. This underestimation was corrected for the automatic measurements at the control for Jan-Jul period in 2017.

For the manual chamber measurements, the fluxes were calculated with nearly the same procedure for the data collected from transects and logging trails. The start and endpoints of the chamber closure were visually identified in the flux calculation. The fluxes were calculated similarly to the automatic chamber measurements so that the specific flux limits for the Gasmet analyser were determined to select between the linear and exponential models as explained in (Korkiakoski et al., 2017). The flux limits for the portable Gasmet system were 35 μg CH_4 m^{-2} h^{-1} and 45 μg N_2O m^{-2} h^{-1} which is about ten times as high as for the LGR and Picarro instruments used in the automatic chamber system due to lower precision of the portable gas analyser.

When using the inside-of-cabin analysers to measure the manual chamber fluxes at the logging trail plots and transects in 2016 and 2017, a 5 min closure time was used at the harvest site. However, at the control transect, a 10 min closure time was used because the N_2O flux was calculated from the data measured with the Gasmet analyser which required a longer closure time to capture the small fluxes more precisely. For the data collected from the control site, the CH_4 flux was calculated by using the first 5 min data acquired from the Picarro analyser. However, N_2O was analysed the same way as the measurements made with the Gasmet analyser described above. When using the automatic chamber system in logging trail plots and transect measurements, the sampled air was not returned to the chamber, causing underpressure inside the chamber and underestimation in the flux estimation. Because the chambers had a vent-tube, we corrected the leakage with an assumption that the underpressure consisted of ambient air.

In this study, a micrometeorological sign convention is used: a positive flux indicates a flux from the ecosystem to the atmosphere (emission) and a negative flux indicates a flux from the atmosphere into the ecosystem (uptake).

Appendix B. Quality control of the flux data

Same filtering methods were used for CH_4 and N_2O fluxes measured with the automatic chamber system as in (Korkiakoski et al., 2017). In short, the hardware problems of the chambers were detected by calculating the normalised root mean square error (NRMSE) of the exponential fit to the CO_2 data measured by Picarro analyser and if NRMSE was > 0.05 the CH_4 and N_2O data were discarded. Also, random spiking was removed by using an iterative standard deviation (± 8 standard deviation) filtering with a 14-day window and one-day time step. At the harvest site, 7.5% of the measured CH_4 fluxes were filtered, and the average number of accepted CH_4 fluxes measured by each chamber was 17427 and 18026 at harvest and control site, respectively. For N_2O fluxes, the filtered percentage was similar to the CH_4 fluxes, but the number of accepted fluxes was slightly larger at the harvest site ($n = 18525$) and markedly lower ($n = 6464$) at the control site. These differences were attributed to different instrument availabilities. The measurements conducted at the logging trail and transect points were checked visually, and only some of the ditch measurements required filtering due to ebullition. The cases of ebullition were visually identified and discarded from further analysis.

Appendix C. Gap-filling and uncertainties of gas balances

The gaps in the automatic chamber data need to be filled to calculate the seasonal and annual balances of CH_4 and N_2O. All the gap-filling procedures described next were conducted separately for each automatic chamber and gas. First, the gaps shorter than one day were filled with linear interpolation and the daily sums were calculated from this gap-filled data. Next, the gaps longer than one day were filled with linear interpolation of the daily sums. From these gap-filled daily sums, the monthly, seasonal, and annual sums were calculated. However, chamber H5 at the harvest site had a three-month gap in 7–10/2015, and this method of gap filling would have probably induced too big of an error in the monthly balances. So, we gap filled the chamber H5 fluxes with linear regression of last three months in 2015 between the chambers H5 and H4 ($r = 0.86$), and H5 and H2 ($r = 0.89$) for CH_4 and N_2O, respectively.

The uncertainties of the gas balances were calculated based on the method in (Korkiakoski et al., 2017) which included: (1) the random error of the linear/exponential fit in flux calculation and (2) the error caused by gap filling. The random and gap-filling errors were combined with the standard accumulation principle of independent errors to get the total error estimate.

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