Emissions of CO_2, N_2O and CH_4 From Cultivated and Set Aside Drained Peatland in Central Sweden

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Northern peatlands are important carbon (C) reservoirs, storing about one-third of the global terrestrial soil C pool. Anthropogenic influences, such as drainage for agriculture and forestry, lower the originally high groundwater level, leading to peat aeration and decomposition. This is particularly reflected in significant losses of CO_2, while fluxes of N_2O and CH_4 are generally considered of minor importance for the overall greenhouse gas (GHG) balance of cultivated peatlands in Scandinavia. Setting land aside from agricultural production has been proposed as a strategy to reduce GHG emissions from drained peatland, restore natural habitats, and increase C sequestration. However, the evidence for this is rather scarce unless drainage is terminated. In this study, we measured respiration using dark automatic chambers, and CO_2, N_2O, and CH_4 fluxes using manual static chambers, on: 1) cultivated peatland and 2) adjacent set-aside peatland in Central Sweden. The set-aside site was found to be a greater source of respiration than the cultivated site, while higher N_2O fluxes and lower CH_4 uptake rates were observed for the cultivated site. However, to compare the full GHG balance and assess the abandonment of drained cultivated peatland, additional measures, such as gross primary production (GPP) but also dissolved organic C losses would have to be taken into account.

Keywords: drained cultivated peatlands, setting land aside, greenhouse gas, chamber measurements, mitigation action, data assimilation

INTRODUCTION

During the Holocene, northern peatlands stored large amounts of carbon (C) (Yu et al., 2009). In a global perspective, they now represent about one-third of the total soil carbon pool (Joosten and Clarke, 2002). Under natural conditions, most peatlands still sequester carbon dioxide (CO_2) from the atmosphere (Gorham 1991), but at the same time emit substantial quantities of C as both CO_2 and methane (CH_4) (Moore and Knowles 1987, 1989). Apart from temperature, the water balance of peat soil is generally regulating decomposition, with decomposition occurring faster in the well-aerated part of the peat above the water table (Silvola et al., 1996; Scanlon and Moore 2000). In several studies, a raised water level has been found to reduce CO_2 and nitrous oxide (N_2O) fluxes, but to increase the emissions of CH_4 (Van de Riet et al., 2013; Regina et al., 2015).

Large areas of peatland are subject to anthropogenic influences and, according to Joosten and Clarke (2002), up to 20% of total global peatland area has been drained for agriculture and forestry. Drained peatlands subside due to oxidation of the organic material, but also due to consolidation,

shrinkage, wind and water erosion and compaction (Eggelsman 1972; Heathwaite et al., 1993; Berglund, 1996). Tillage and fertilization can increase decomposition rates caused by higher eration (Glenn et al., 1993; Maljanen et al., 2007). Drainage, in combination with the naturally high organic C concentration in peat soil, leads to increased C losses in the form of CO_2 emissions (e.g., Joosten and Clarke, 2002; Lohila et al., 2004; Joosten 2009; Frolking et al., 2011; Salm et al., 2012; Ballantyne et al., 2014). Moreover, these highly elevated CO_2 emissions persist for decades after drainage of peat soil (e.g., Koizumi et al., 1999; Lohila et al., 2003). The global potential of peatland restoration in reducing GHG emissions is therefore estimated to be very high, 0.31–3.38 Gt CO_2-equivalents according to Leifeld and Menichetti (2018).

In Sweden, peatlands of varying thickness cover approximately 25% of the land area (Fredriksson 1996). Around 8.6% of Swedish agricultural soils are classified as agricultural organic soils (peat and gyttja) (Berglund and Berglund, 2010), which are considered to be 'hotspots' of greenhouse gas (GHG) emissions. Despite comprising a small proportion of total agricultural area, cultivated organic soils in Sweden and Finland are considered to be major sources of CO_2 and N_2O emissions (Kasimir-Klemendtsson et al., 1997; Maljanen et al., 2010). Enhanced nutrient availability through fertilizer application and drainage has been shown to increase N_2O emissions (Alm et al., 2007; Teh et al., 2011; van Beek et al., 2011; Salm et al., 2012). Emissions of CH_4 have been found to decrease due to soil eration and management, suggesting that cultivated peat soils may even act as CH_4 sinks (Kasimir-Klemendtsson et al., 1997; Maljanen et al., 2003; Smith and Conen 2004; Maljanen et al., 2010; Schrier-Uijl et al., 2014; Knox et al., 2015; Wang et al., 2017). Because of their comparatively low fluxes, N_2O and CH_4 are reported to play only a minor role in the overall GHG balance of cultivated peat soils in Northern Europe (e.g., Maljanen et al., 2004; Grönlund et al., 2006; Berglund and Berglund, 2010; Karki et al., 2015). However, most studies agree that conversion of wetlands for agriculture leads to significant net losses of CO_2. Lamb et al. (2016) suggested that land sparing, i.e., setting land aside from agriculture, could have the potential to reduce GHG emissions from agricultural land, through concentrating crop production to intensively managed sites, while restoring natural habitats on the abandoned land and increasing C sequestration. Permanent vegetation on abandoned peatland sites would provide year-round soil cover and enhanced photosynthesis compared with seasonal agricultural crops. This could partially compensate for C losses induced e.g., by the lower water table (Hendriks et al., 2007; Shurpali et al., 2009). However, information about the effects of abandoning or setting aside drained peatlands from agricultural use is scarce. In a study in Newfoundland, Wang et al. (2018) found that an abandoned peatland pasture sequestered more CO_2 from the atmosphere than an adjacent undisturbed bog. On including emissions of N_2O and CH_4, Wang et al. (2020) found that the same abandoned pasture had a cooling effect, primarily resulting from changes in CO_2 and CH_4, while the bog acted as nearly climate-neutral. Based on these findings, the authors reiterated the suggestion that abandoning cultivated

peatland sites could mitigate CO_2 emissions. In contrast, Maljanen et al. (2007) found that abandoned organic soils in Finland showed slight net CO_2 uptake during the growing season, but were CO_2 sources outside that period, and that mean annual net CO_2 emissions from those soils were close to those from cultivated cropland. Similarly, Hadden and Grelle (2017) found that a Swedish peatland site under cultivation was a small net C sink, while a previously cultivated site that had been abandoned for 25 years was a C source.

The aim of the present study was to assess the emissions of CO_2, N_2O, and CH_4, from both a cultivated and a set aside organic soil. The study was conducted at two neighboring sites in a peatland area in Central Sweden that was drained in 1878 for agricultural use (Nerman 1898). One of these sites was set aside in the early 1990s, which means that agricultural management has been stopped. However, this did not involve closing the ditches or rising the water table. The other site is still used for agricultural crop production, mainly spring cereals.

MATERIALS AND METHODS

Study Site

Two experimental sites, cultivated (Cult) and set-aside (SA) were established at the margin of a natural peatland area about 30 km north of Uppsala in Central Sweden (**Figure 1**). The climate in this area is cold-temperate (Dfb according to the Köppen-Geiger classification) with mean annual temperature of 5.6°C and mean annual precipitation of 544 mm.

The area underwent large-scale drainage during lowering of Lake Tämnaren in 1878 (Nerman 1898), which also affected the two study sites. The cultivated site (60.0835°N, 17.233°E) was cropped with spring cereals during the study period (2013–2016), to produce standing forage for wild birds. The crop was not harvested, but the site was tilled and re-sown every spring. Prior to the study period, annual crops had been grown in rotation with grass leys (4 out of 10 years) supplying forage for dairy cows. Manure was applied and complemented with small amounts of nitrogen fertilizer during that period. The set-aside site (60.079°N, 17.236°E) was established 0.5 km south of the cultivated site and had been set aside (under permanent grassland) for more than 20 years prior to the start of the study. During the study period, the grass growing at the site was occasionally cut once a year and left on the field.

Soil properties were analyzed at the two sites (**Table 1**). Total nitrogen (N_{tot}) and total carbon (C_{tot}) content were determined by dry combustion on a LECO CN-2000 analyzer (St. Joseph, MI, USA). Soil pH and electrical conductivity (EC) were measured at a soil:solution ratio of 1:5 with deionized water. Four replicate undisturbed soil cores (7.2 cm diameter, 10 cm high) from each soil depth were used for determination of dry bulk density and volumetric water content at suction heads of 0.05, 0.3, 0.6, 1.0 and 6.0 m water column (approximately 0.5, 3.0, 6.0, 10.0, and 60.0 kPa). Physical wilting point (water content at 150 m water column) and particle density were determined on disturbed soil samples. Porosity was calculated from particle and dry bulk densities. Shrinkage was not considered when calculating water content at
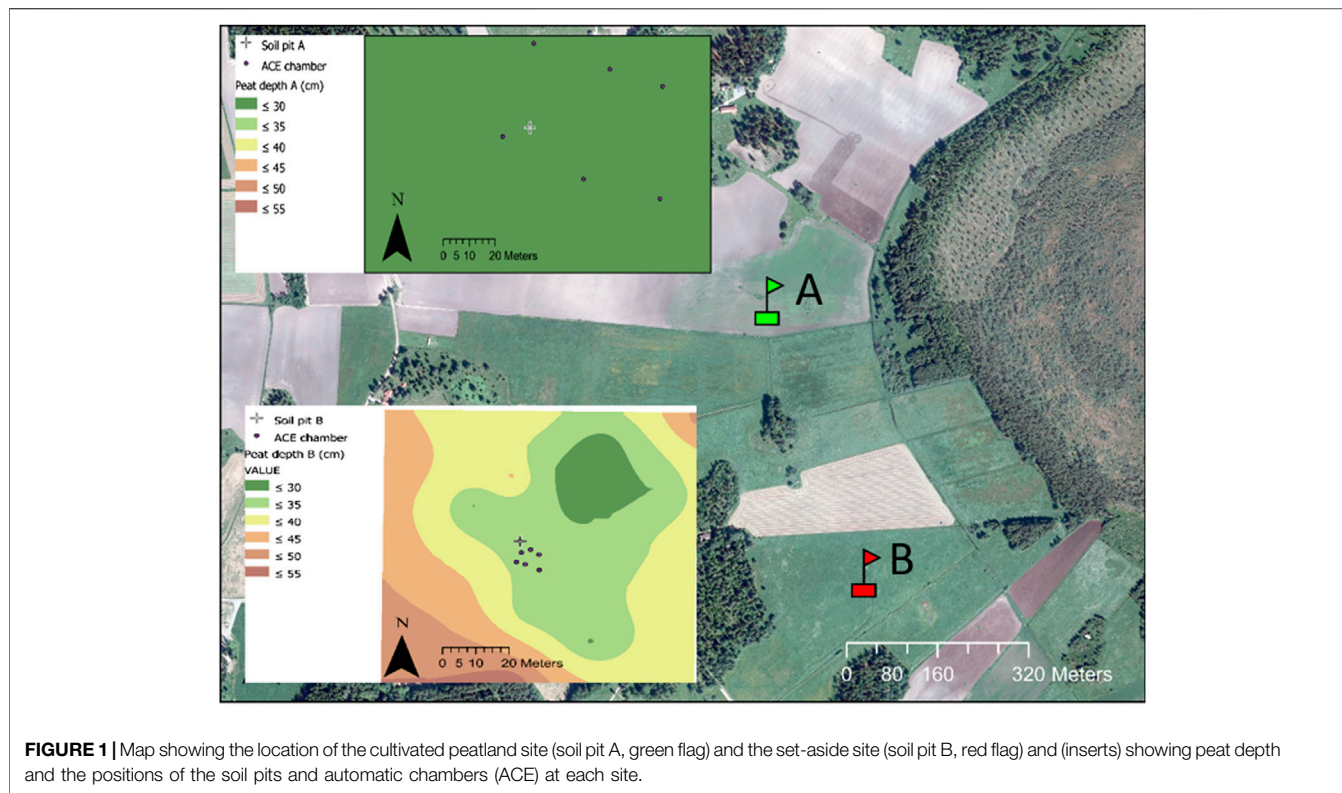
FIGURE 1 | Map showing the location of the cultivated peatland site (soil pit A, green flag) and the set-aside site (soil pit B, red flag) and (inserts) showing peat depth and the positions of the soil pits and automatic chambers (ACE) at each site.

different tensions. Saturated hydraulic conductivity was determined using a constant head method (unit hydraulic gradient) on another set of undisturbed soil cores (4 replicates). Humification degree (H1-H10) of the peat was determined according to von Post (1922). The water table was 0.5 m below the soil surface at both sites at the time of soil sampling in October 2014.

Climate Data

Abiotic variables measured included air temperature, precipitation, and global radiation, which were taken from the nearest official meteorological station (Kerstinbo; 60.270°N, 16.980°E).

Gas Sampling Set-Up

Measurements were performed every year between 2012 and 2016. During that time, gas measurements using dark chambers were taken in 2012, 2013, 2015, and 2016, while CO_2 measurements with automatic chambers were taken in 2013, 2014, and 2015. The duration and intensity of measurements varied between the years.

To estimate root respiration at the set-aside site, the grass sward was removed (SA-bare) and the emissions were compared with those from vegetated plots (SA-veg).

Manual Measurements

Manual measurements were made using the closed chamber method (e.g., Flessa et al., 1995) and following Norberg et al.

TABLE 1 | Soil properties at the set-aside and cultivated study sites. Soil sampling was performed on bare soil at the set-aside site, after removing the grass sward. All values are means (n = 4). H10 = humification degree (von Post, 1922), C_{tot} = total carbon content, N_{tot} = total nitrogen content, soil pH was measured at a soil:solution ratio of 1: 5 with deionized water, EC = electric conductivity, ρ_b = dry bulk density, K_{sat} = saturated hydraulic conductivity, n.a. = no measurements available.

Depth [cm]	Soil type	C_{tot} [%]	N_{tot} [%]	C/N [-]	pH [-]	EC [$\mu S\ cm^{-1}$]	ρ_b [g cm^{-3}]	Porosity [vol%]	K_{sat} [cm h^{-1}]
					Set-aside				
5–15	Fen peat (H10)	33.6	2.03	16.5	5.6	37.2	0.40	78	138
20–30	Fen peat (H10)	33.3	1.97	16.9	5.5	33.4	0.39	78	83
35–45	Gyttja clay	0.90	0.06	15.0	6.3	85.8	0.99	64	3.6
50–60	Clay	0.81	0.06	13.5	6.9	209	n.a	n.a	n.a
					Cultivated				
5–15	Fen peat (H10)	31.7	1.93	16.4	5.7	95.9	0.53	71	2
25–35	Gyttja clay	3.57	0.36	9.9	5.7	48.1	0.89	66	40
40–45	Clay	n.a	0.04	n.a	7.6	270	n.a	n.a	n.a

(2016a, 2016b). In brief, at each site six PVC rings with a base area of 0.07 m^2 were inserted into the soil to approximately 5 cm depth, to act as anchors for the chambers. To measure soil-atmosphere exchange, opaque chambers (V = 0.009 m^3) were set on the rings and sealed with rubber seals. On plots with vegetation, the crop was cut prior to the measurements. Soil and plant (mainly root) respiration was measured in two ways: 1) using a portable infrared analyzer, where measurements were performed every 5 s over a period of 3–5 min (Norberg et al., 2016b); and 2) together with N_2O and CH_4, using vial sampling and gas chromatography. For days when both methods were used, CO_2 fluxes are presented as the mean of both methods. For N_2O and CH_4, the headspace was sampled by circulating the air between the chamber and 22 ml vials for approximately 30 s (Norberg et al., 2016a). Gas samples were taken at 0, 10, 20, and 30 min after closure, and analyzed in the laboratory using gas chromatography (Perkin Elmer Clarus 500, USA). Gas fluxes were calculated using the HMR approach (Pedersen et al., 2010), which is based on the model of Hutchinson and Mosier (1981). Depending on the trend in the concentration data over time, the fluxes were calculated by either a linear or non-linear function. "No flux" is recommended by the algorithm when the concentration data are noisy and do not show a clear trend. In our dataset, this was the case for 25% of the N_2O fluxes and 21% of the CH_4 fluxes, and those were set to zero. In total, 18% and 38% of the fluxes were calculated using the non-linear function for N_2O and CH_4, respectively, while for the remaining fluxes a linear function described the trend best.

Soil moisture (volumetric water content) and soil temperature at 5 cm depth were measured next to each chamber at each measurement event, using a WET sensor (Delta-T) connected to a HH2 moisture meter read-out unit. The air-filled pore space (AFPS) was calculated using the volumetric water content and soil porosity.

Automatic Measurements

Continuous observations of soil and plant respiration (vegetated plots, SA-veg), and of peat oxidation (vegetation cover and roots removed; bare soil, SA-bare), were made using Automatic Soil CO_2 Exchange (ACE) stations (ADC BioScientific Limited, Hertfordshire, United Kingdom). Each station consists of an electronic box and a motorized moveable arm that contains an infrared gas analyzer. Besides measurements of soil respiration, the stations were used to record soil temperature at 5 and 10 cm depth (range: −10° to 50°C), and moisture at 5 cm depth measured as voltage (range 0–2 V), which was converted to volumetric water content. The area of soil analyzed was defined by a stainless steel soil collar (23 cm in diameter) which was inserted into the soil to a depth of 5 cm. To perform CO_2 measurements, the collar was automatically covered by a sealed lid enclosing a volume of 2.6 L. Internal flux was estimated based on the change in CO_2 concentration in the chamber during the closure time, by fitting either a linear or a non-linear function to the concentration data. After measurement, the lid was automatically removed from the collar and the system was left open to the environment. Measurements were performed at hourly intervals. The closure

time depended on the change in concentration (maximum 10 µmol m^{-3}), but was never longer than 5 min.

Statistical Analysis

The data were analyzed using the R software (R i386 3.3.1, R Core Team 2016). For data processing, the *openxlsx* package (Schauberger and Walker, 2019) and the ddply function (*plyr* package, Wickham 2011) were used. Gas fluxes were calculated using the *HMR* package (Pedersen 2017). Differences between the sites and years were determined by analysis of variance (ANOVA) (Type-III error, *car* package; Fox and Weisberg 2019) and tested by Honest Significant Difference test (HSD.test in the *agricolae* package; de Mendiburu 2017). Differences were considered significant at $p < 0.05$. Diagrams were created using the *ggplot2* (Wickham 2009) and *cowplot* (Wilke 2017) packages. Pearson correlation coefficient (r) was used to determine linear correlations between manually measured gas fluxes and simultaneously measured soil temperature, and between manually measured gas fluxes and the interaction between soil temperature and water content (calculated as soil temperature [°C] x water content [%]). Outlier analysis was carried out using the *outliers* package (Komsta 2011).

RESULTS AND DISCUSSION

The peat layer (average of nine measurements) was shallower at the cultivated site (25 cm) than at the set-aside site (32 cm). The shallower peat layer at the cultivated site may be due to initial differences in peat depth or cultivation, which may have caused erosion and/or faster decomposition of the peat. It can be assumed that the original peat layer, i.e., before drainage of the area in 1878 (Nerman 1898), was much thicker and that the peat surface was at higher elevation. Subsidence rates due to agricultural drainage of peat soils in Nordic conditions can vary from 0.5 cm yr^{-1} on fields with pasture to 2.5 cm yr^{-1} on fields with row crops (Berglund 1996). Subsidence rates in this range (1–3 cm yr^{-1}) have been reported by e.g., Armentano (1980). The proportion of subsidence attributable to oxidation is estimated to range between 13 and 90% (Armentano and Menges 1986; Joosten and Clarke, 2002). For Swedish conditions, a value of 35% has been used as an average for all cultivated peat soils (Berglund and Berglund, 2010). Varying subsidence rates of fields under different crops have been taken as an indication that GHG emission rates vary due to the crop grown (Kasimir-Klemendtsson et al., 1997). In the present study, this would mean much higher emissions from the cultivated field compared with the set-aside field with permanent grass vegetation.

Manual Measurements
CO_2

The measured CO_2 fluxes varied between sites and years. Median fluxes (2012–2016) were significantly higher from SA-veg and SA-bare (743 and 791 mg m^{-2} h^{-1}, respectively) than from the cultivated site (415 mg m^{-2} h^{-1}) (**Tables 2** and **3**). No significant

TABLE 2 | Ecosystem respiration, nitrous oxide (N_2O) and methane (CH_4) fluxes measured manually at the set-aside and cultivated sites between 2012 and 2016. Values shown are median (Med), minimum (Min), and maximum (Max). n = number of data points. GWP = Global Warming Potential.

Treatment	CO_2 [mg m^{-2} h^{-1}]				N_2O [µg m^{-2} h^{-1}]				CH_4 [µg m^{-2} h^{-1}]				GWP[a] [mg m^{-2} CO_2-eq]
	Med	Min	Max	n	Med	Min	Max	n	Med	Min	Max	N	
Set-aside (SA) Vegetated (veg)													
2012	1,192	307	2,547	48									
2013					−9	−458	67	36	0	−778	237	36	
2015	401	144	1,098	58	7	−474	503	129	−37	−221	67	128	
2016	1,018	224	1999	54	0	−295	354	48	−56	−345	69	48	
Average[a]	743	144	2,547	160[b]	0	−474	503	213[b]	−36	−778	237	212[b]	742
Bare soil (bare)													
2012	880	192	4,102	54									
2015	189	98	561	10	0	−81	85	24	−33	−166	11	24	
Average[a]	791	98	4,102	64[b]	0	−81	85	24[b]	−33	−166	11	24[b]	790
Cultivated													
2012	560	69	1,661	72									
2013	543	227	871	30	3	−156	719	36	0	−56	545	36	
2015	226	43	823	84	63	−128	2,447	151	−7	−239	132	153	
2016	645	170	1,285	53	0	−207	303	48	0	−373	84	48	
Average[a]	415	43	1,661	239[b]	30	−207	2,447	235[b]	−5	−373	545	237[b]	424

[a]*Median over the study period.*
[b]*Total number of data points.*
[c]*Calculated using the GWP of N_2O (298) and CH_4 (34) for a 100-year period (Myhre et al., 2013).*

TABLE 3 | Differences in greenhouse gas (GHG) fluxes determined by manual measurements between the set-aside (SA) and cultivated sites and between years, based on analysis of variance (ANOVA) and Honest Significant Difference test. Different letters indicate significant differences between sites/years.

	CO_2 [mg m^{-2} h^{-1}]	N_2O [µg m^{-2} h^{-1}]	CH_4 [µg m^{-2} h^{-1}]
Site	a	a	a
SA-veg	A	B	B
SA-bare	A	B	B
Cultivated	B	A	A
Year	a	n.s	a
2012	A		n.a
2013	B		A
2015	B		B
2016	A		B

[a]*Significant difference (p < 0.05). n. s = no significant difference. n. a. = no data available.*

TABLE 4 | Season-specific average plant-derived carbon dioxide (CO_2) flux [mg m^{-2} h^{-1}] at the set-aside site in seasons covered by manual measurements in 2012 and 2015.

Season	Average CO_2 flux [mg m^{-2} h^{-1}]			
	Vegetated	Bare soil	Plant-derived [%]	n
Spring (until 22 June)	896 b	542 ab	40	2
Summer (until 15 August)	1,118 a	864 a	23	6
Autumn (until 1 November)	398 c	215 b	46	9

n = number of data points. Different letters after values indicate significant differences between seasons, based on analysis of variance (ANOVA) and Honest Significance Difference test.

differences between SA-veg and SA-bare were found (**Table 3**). According to the IPCC (2013) emission factors, CO_2 emissions should be much lower fro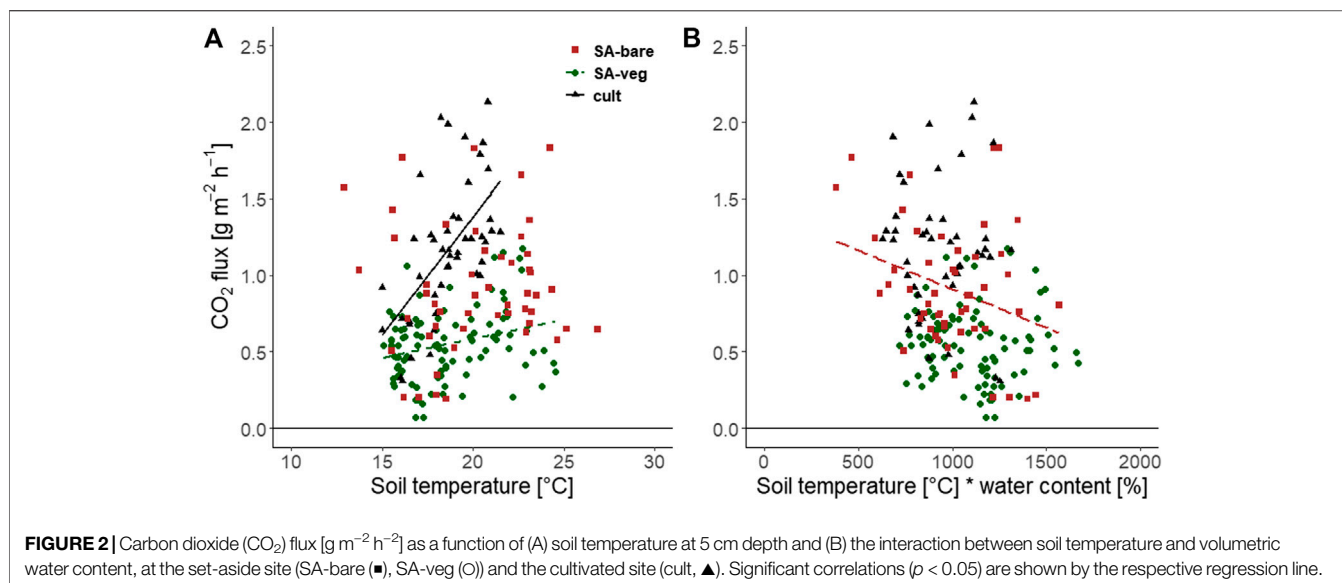m the set-aside site (grassland, drained, boreal: 5.7 t CO_2-C ha^{-1} yr^{-1}) compared with the cultivated site (cropland, drained, boreal: 7.9 t CO_2-C ha^{-1} yr^{-1}), but this was not confirmed by our data.

Plant-derived CO_2, estimated as the difference between CO_2 fluxes from simultaneously measured vegetated plots and bare soil, was 37% (**Table 4**). This is within the range reported by Norberg et al. (2016b) (27%) and Berglund et al. (2011) (27–63%) for different cultivated and non-cultivated organic soils in southern and central Sweden. Berglund et al. (2011) identified seasonal variation in the plant contribution to CO_2 emissions, with the lowest contribution in spring (27%) and the highest in autumn (57%). Based on our limited dataset (17 data points in total), average plant-derived CO_2 fluxes were lower in summer than in spring and autumn (**Table 4**). This can be explained by the significantly higher CO_2 fluxes from both SA-veg and SA-bare during summer ($p < 0.01$).

Soil temperature and soil eration status had a strong impact on formation and release of CO_2, and emissions increased with increasing temperature (**Figure 2**). The correlation with temperature was significant for SA-veg (r = 0.60) and the cultivated site (r = 0.27), but not for SA-bare (r = 0.16). The increase in CO_2 fluxes with increasing air-filled pore space (AFPS) in the soil was significant only for the bare soil (r = 0.45) (data not shown). The relationship between CO_2 emissions and the interaction of soil temperature and water content (soil temperature x water content) was also significant only for SA-bare (r = -0.31).

N_2O

Median fluxes of N_2O from the cultivated site (30 µg m^{-2} h^{-1}) were significantly higher than from the set-aside site (**Table 2**). The fluxes from SA-veg were on average negative in 2013, but were positive in 2015 and zero in 2016. The bare soil (SA-bare)

FIGURE 2 | Carbon dioxide (CO_2) flux [g m^{-2} h^{-2}] as a function of (A) soil temperature at 5 cm depth and (B) the interaction between soil temperature and volumetric water content, at the set-aside site (SA-bare (■), SA-veg (O)) and the cultivated site (cult, ▲). Significant correlations ($p < 0.05$) are shown by the respective regression line.

showed zero N_2O fluxes on average. This was mainly due to zero fluxes during summer and autumn, while fluxes were negative during spring ($-39 \mu g m^{-2} h^{-1}$; **Table 5**). In SA-veg, the N_2O fluxes were similar in spring and autumn (7 and 5 $\mu g m^{-2} h^{-1}$, respectively) and were slightly, but not significantly, higher in summer (14 $\mu g m^{-2} h^{-1}$). At the cultivated site, N_2O fluxes were not significantly different during spring and summer (141 and 288 $\mu g m^{-2} h^{-1}$, respectively), but were significantly lower ($p < 0.05$) in autumn (27 $\mu g m^{-2} h^{-1}$).

In general, N_2O fluxes showed large temporal and spatial variation and both consumption and emission were observed at all sites, irrespective of the vegetation. Absence of plants could be assumed to result in more available nitrogen (N) and, consequently, higher N_2O emissions, due to lack of N uptake by plant roots. However, we found that vegetated plots acted as N_2O sources, while fluxes were on average negative for bare soil plots (**Table 2**). Johansson et al. (2011) found a negative correlation between N_2O consumption and nitrate (NO_3^-) concentration, suggesting that consumption is linked to NO_3^- deficiency (lack of electron acceptors) and that denitrifying bacteria use N_2O as a substitute. Although it

can be assumed that denitrification was the main process for N_2O production at all sites in this study, the negative fluxes found for SA-bare indicate reduction of N_2O to molecular nitrogen (N_2) in the final step of the denitrification process. Based on our dataset, decreasing soil moisture, and thus increasing AFPS, significantly decreased N_2O emissions at the cultivated site ($r = -0.25$). Furthermore, we found no relationship with soil temperature or with the interaction between soil temperature and soil moisture at either SA-veg or the cultivated site, which suggests that other factors influence the formation and emission of N_2O from the soil (**Figure 3**). Although crop management was quite extensive at the cultivated site, its history of manure application and the consequences of tillage in late spring, i.e., before the measurements started, might still be visible in the observed N_2O emissions.

CH_4

Both the cultivated and the set-aside site were found to be a sink for CH_4 and the latter acted as a sink irrespective of the vegetation (**Table 2**). In 2013, average fluxes were zero from both the set-aside and the cultivated site. In contrast, there was net uptake of CH_4 in 2015 on both sites and in 2016 on the set-aside site. Small and mostly negative fluxes from drained peat soils have been observed in many studies in Sweden (e.g., Kasimir-Klemedtsson et al., 2009; Berglund and Berglund, 2010), Finland (e.g., Nykänen et al., 1995; Maljanen et al., 2003), Canada (Glenn et al., 1993), and Estonia (Salm et al., 2012).

In the present study, CH_4 uptake rates were significantly higher at the set-aside site (36 $\mu g m^{-2} h^{-1}$) than at the cultivated site (5 $\mu g m^{-2} h^{-1}$). Although CH_4 uptake by SA-bare (33 $\mu g m^{-2} h^{-1}$) was on average only slightly lower than that by SA-veg, the latter showed greater variation, with both high emissions (237 $\mu g m^{-2} h^{-1}$) and high uptake rates (778 $\mu g m^{-2} h^{-1}$). The function of all sites as zero emissions or even a net sink

TABLE 5 | Season-specific median nitrous oxide (N_2O) flux [$\mu g m^{-2} h^{-1}$] from the set-aside site (SA-bare and SA-veg) and the cultivated site.

Season	Median N_2O flux [$\mu g m^{-2} h^{-1}$]			
	Set-aside		Cultivated	n
	Vegetated	Bare soil		
Spring (until 22 June)	7	−39 b	141 a	5
Summer (until 15 August)	14	0 a	288 a	4
Autumn (until 1 November)	5	0 ab	27 b	6

Only measurements taken simultaneously, i.e., on the same day, are shown. n = number of data points. Different letters after values indicate significant differences between seasons, based on analysis of variance (ANOVA) and Honest Significant Difference test.
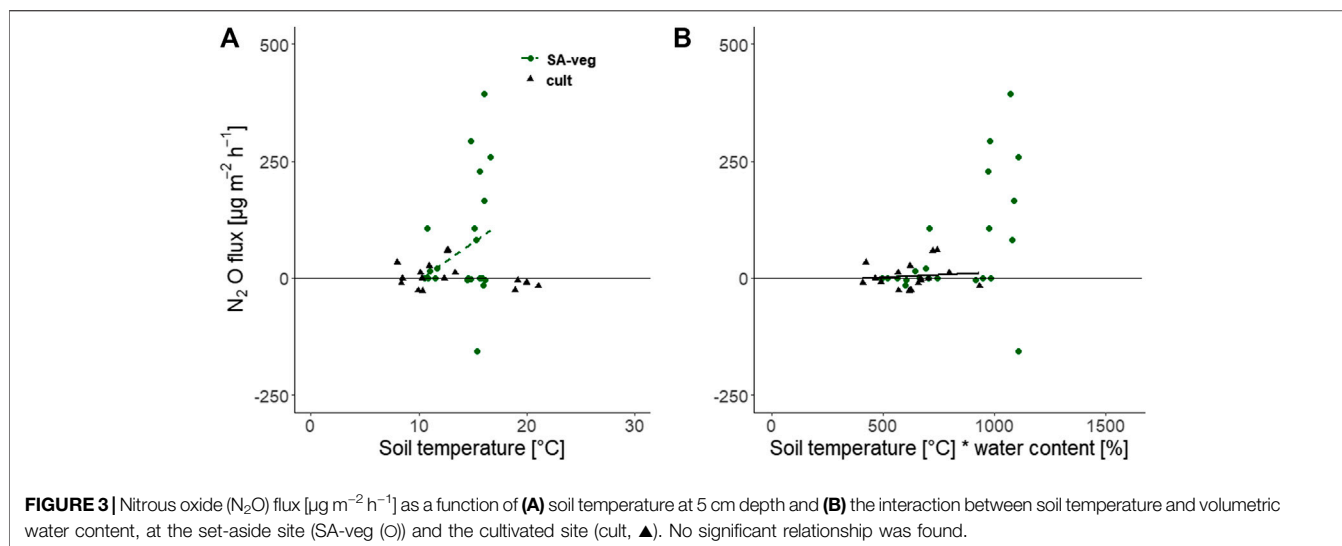
FIGURE 3 | Nitrous oxide (N_2O) flux [μg m^{-2} h^{-1}] as a function of **(A)** soil temperature at 5 cm depth and **(B)** the interaction between soil temperature and volumetric water content, at the set-aside site (SA-veg (O)) and the cultivated site (cult, ▲). No significant relationship was found.

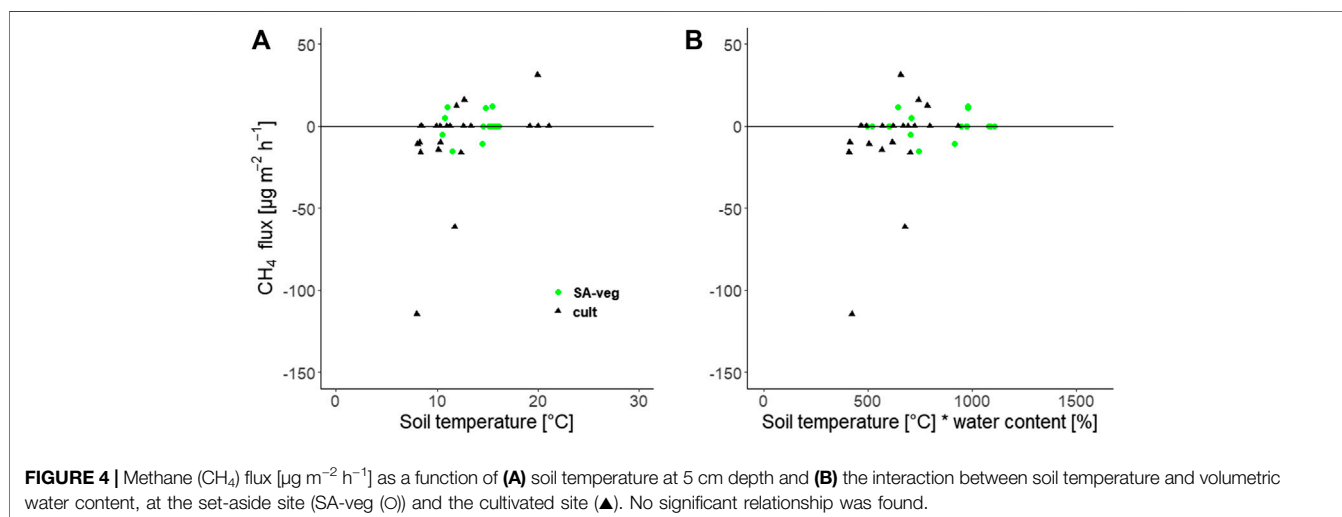**FIGURE 4 |** Methane (CH_4) flux [μg m^{-2} h^{-1}] as a function of **(A)** soil temperature at 5 cm depth and **(B)** the interaction between soil temperature and volumetric water content, at the set-aside site (SA-veg (O)) and the cultivated site (▲). No significant relationship was found.

for CH_4 is likely to be related to the shallow peat layer, soil drainage, and the associated lowering of the water table. In the presence of O_2, CH_4 is consumed by methanotrophs for energy production and growth (Lai 2009). Methanotrophic activity in peatland soils is highest at shallower depths and in the boundary between the aerobic and anaerobic zones, where the ratio of CH_4 to O_2 is optimal (Dedysh 2002; Lai 2009). Consequently, rising water levels, e.g., as a consequence of poor drainage, have been found to transform peat soils from a net sink into a net source of CH_4 (e.g., Regina et al., 2007, 2015). The peat soil at the set-aside site was less compacted than that at the cultivated site, with lower dry bulk density and higher porosity and saturated hydraulic conductivity (**Table 1**). This difference in aeration and oxidizing conditions may explain the higher uptake of CH_4 at the set-aside site compared with the cultivated site.

We found no significant correlation between CH_4 flux and soil temperature (**Figure 4A**). However, CH_4 fluxes decreased with increasing AFPS in both SA-bare and the cultivated site

TABLE 6 | Season-specific median methane (CH_4) flux [μg m^{-2} h^{-1}] from the set-aside site (bare soil and vegetated) and the cultivated site.

Season	Median CH$_4$ flux [μg m^{-2} h^{-1}]			
	Set-aside		Cultivated	n
	Vegetated	Bare soil		
Spring (until 22 June)	−29	−35	−11	6
Summer (until 15 August)	−27	−26	−5	7
Autumn (until 1 November)	−44	−48	−9	11

n = number of data points. Fluxes did not differ significantly between the seasons.

(data not shown), but this relationship was not significant at either of the sites. Similarly, Kasimir-Klemedtsson et al. (2009) found positive CH_4 fluxes in wetter areas, and CH_4 uptake in drier areas, of a cultivated Histosol in southern Sweden. In agreement with Kasimir-Klemedtsson et al. (2009), we did not find a significant correlation with soil temperature or with the
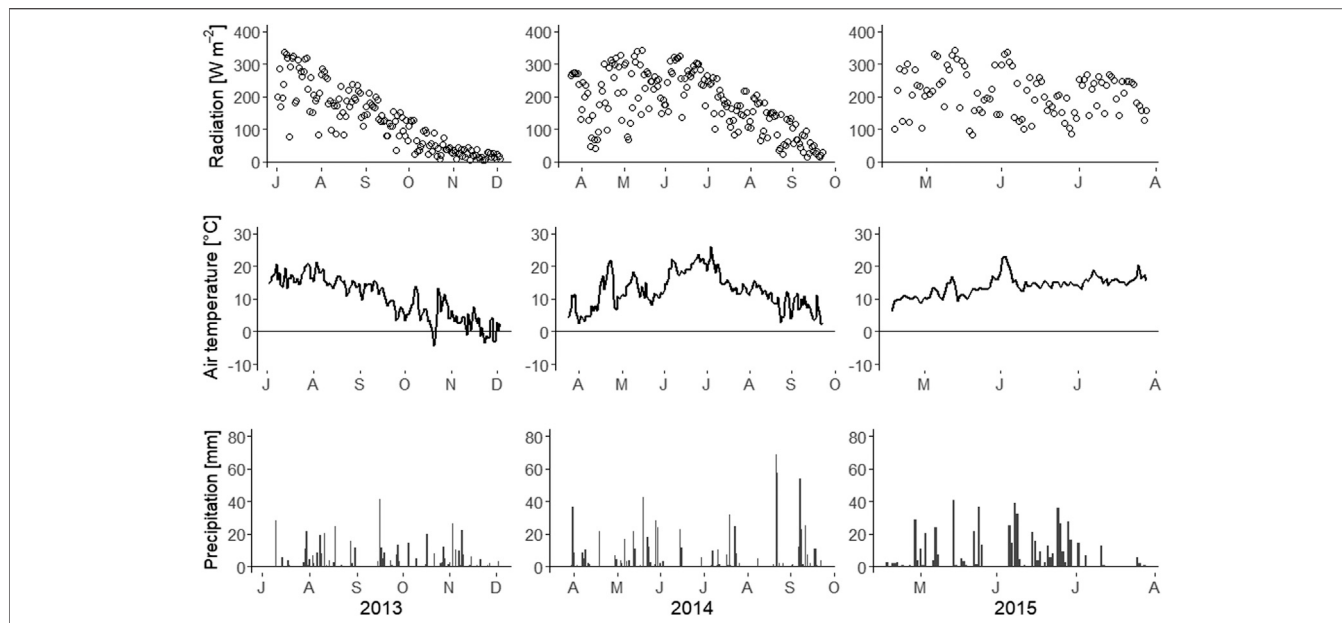
FIGURE 5 | Daily global radiation **(top)**, air temperature **(middle)**, and precipitation **(bottom)** in 2013, 2014, and 2015, measured at the nearest official meteorological station (Kerstinbo) to the study site.
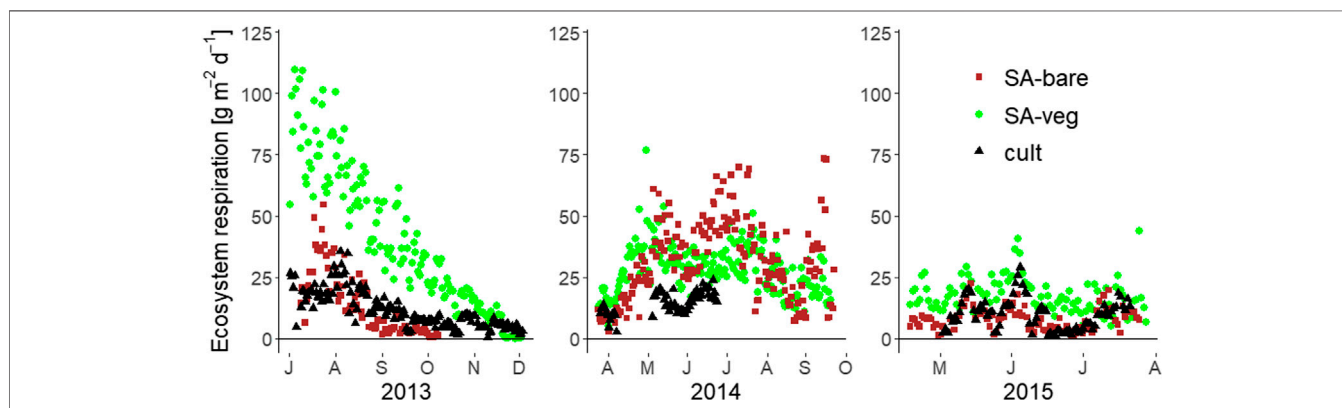

FIGURE 6 | Daily ecosystem respiration [g m^{-2}] measured by automatic chambers at the set-aside site (SA-bare and SA-veg) and the cultivated site (cult) in July-December 2013, April-October 2014, and May-August 2015.

interaction between soil temperature and water content. Lack of correlation between CH_4 flux and soil parameters may be explained by low emissions levels and fluctuations around zero (Kasimir-Klemedtsson et al., 2009).
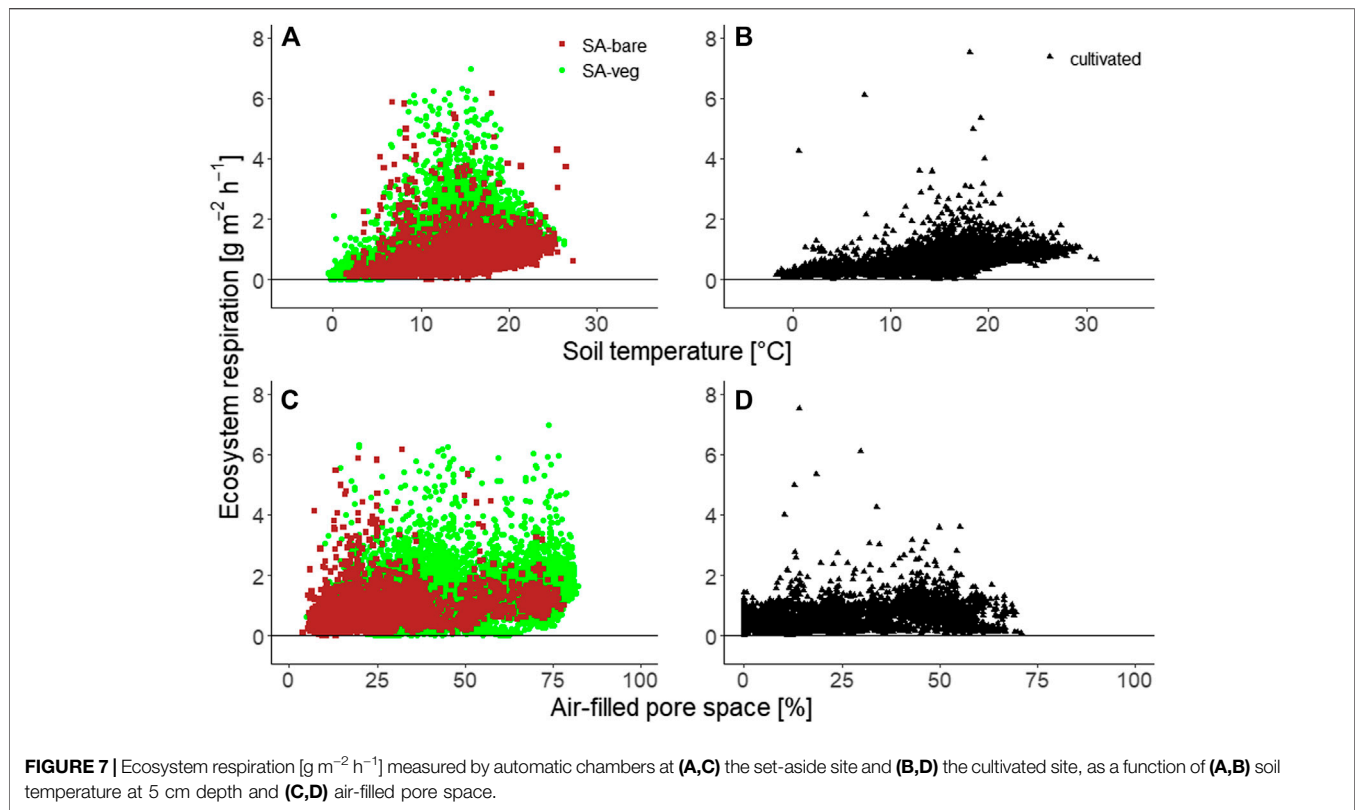
Using the eddy covariance technique at the set-aside site, Hadden and Grelle (2017) observed higher CH_4 uptake rates during the summer months, while spring and winter usually had small but positive fluxes. In general, those authors found that CH_4 exchange rates were close to zero for air temperatures below freezing, and that higher soil temperatures led to higher uptake rates. On comparing manual measurements made simultaneously in all study plots, we did not find any significant differences between the seasons (**Table 6**). However, uptake rates at the set-aside site were on average highest in autumn (irrespective of

TABLE 7 | Daily ecosystem respiration [mg m^{-2} h^{-1}] in July, determined using automatic chambers, in 2013 (n = 23), 2014 (n = 24), and 2015 (n = 30).

Year	Median CO$_2$ flux [mg m^{-2} h^{-1}]		
	Set-aside		**Cultivated**
	Vegetated	**Bare soil**	
2013	1881 a	1094 a	901 a
2014	1585 b	867 b	908 a
2015	1008 c	632 c	390 b

Flux estimates are based on measurements taken on the same days in July in each year. Values followed by different letters indicate significant differences between years (p < 0.05).

vegetation), while uptake rates at the cultivated site were highest in spring and similar in summer and autumn (**Table 6**).

FIGURE 7 | Ecosystem respiration [g m^{-2} h^{-1}] measured by automatic chambers at **(A,C)** the set-aside site and **(B,D)** the cultivated site, as a function of **(A,B)** soil temperature at 5 cm depth and **(C,D)** air-filled pore space.

Automatic Chambers

Respiration measurements using automatic chambers were made between 2013 and 2015. Comparing the three years based on the month of July, which was covered in all three years, the average temperature was higher in 2014 (20°C) than in 2013 (17°C) and 2015 (15°C). However, the summers in 2013 and 2014 were rather dry, with rainfall in July of 25 and 16 mm, respectively. In contrast, 2015 was characterized by a wetter summer, with 105 mm of rain in July (**Figure 5**).

Measured CO_2 followed a seasonal trend, with higher emissions during the summer months and decreasing emissions as winter approached (**Figure 6**).

In July, CO_2 fluxes were higher from the vegetated set-aside plots than from the bare set-aside plots or the cultivated site (**Table 7**). One explanation for this difference might be higher C availability at the set-aside site due to the thicker peat layer, slightly higher soil C content, and greater biomass of vegetation compared with the cultivated site (Hadden and Grelle 2017).

Soil respiration from the bare soil plots (859 mg m^{-2} h^{-1}) was around 55% of ecosystem respiration (vegetated plots, 1,575 mg m^{-2} h^{-1}) when averaged over all three years. In individual years, soil respiration was 45% (2013), 52% (2014), and 37% (2015) of ecosystem respiration. These estimates are lower than the values obtained using the manual chambers, according to which soil respiration accounted for 63% of ecosystem respiration (total CO_2 flux). Similar estimates (62%) were made by Lohila et al. (2003) for a Finnish peat

soil in the middle of the summer. Measurements made within controlled lysimeter experiments with ryegrass on a similar peat soil estimated the contribution of soil respiration to total summer-time CO_2 emissions to be 46% (Berglund et al., 2011). Our values are also slightly lower than estimates made in other studies using eddy covariance and chambers, according to which soil respiration accounted for 68–80% of total ecosystem respiration (Goulden et al., 1996; Law et al., 1999; Granier et al., 2000; Davidson et al., 2002). Generally, these rates typically vary throughout the year and can be expected to be lower during the growing season and higher during the non-growing season. For example, Bolstad et al. (2004) found that soil respiration was more than 60% of total ecosystem respiration during the growing season and more than 90% during the non-growing season. However, these estimates relate to forests and forest soils, respectively. The proportion of bare soil respiration estimated in this study (55%) relates to the growing season and the month of July. Moreover, it only relates to the set-aside site, since based on the data available we were unable to make an estimate for the cultivated site. The generally lower respiration observed at the cultivated site might be due to e.g., the shallower peat layer and lower root biomass, leading to less heterotrophic and autotrophic respiration, respectively.

As expected, the variation in ecosystem respiration over single days was rather moderate, but average fluxes increased slightly throughout the morning and decreased toward the evening. In general, measured fluxes were higher at higher

FIGURE 8 | Ecosystem respiration [g m^{-2} h^{-1}] as a function of both soil temperature [˚C] and air-filled pore space [AFPS, %]. Individual squares represent increments of 2˚C and 5% AFPS, and colors represent the median CO2 fluxes.
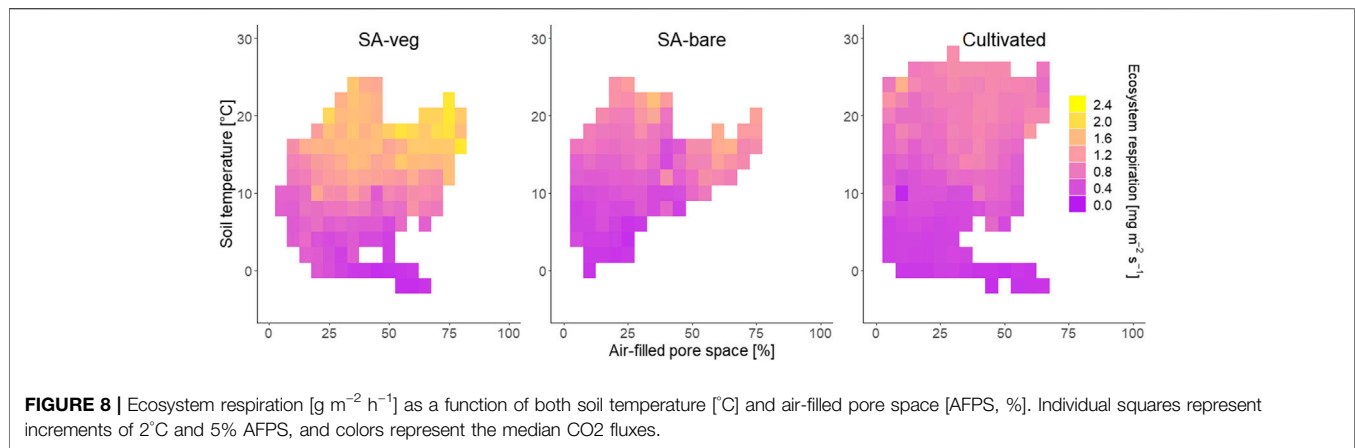
soil temperatures (**Figures 7A,B**). No such clear correlation was seen for changes in AFPS (**Figures 7C,D**). The fluxes determined using the automatic chambers are in good agreement with the respective fluxes captured by eddy covariance at the same two sites and over the same period (Hadden and Grelle 2017; data not shown).

Considering both environmental factors simultaneously revealed that higher fluxes generally occurred at higher soil temperatures (**Figure 8**). At all sites, this was also related to higher AFPS but, due to the magnitude of the emissions, this effect was less visible for the cultivated site. The highest average emissions over the respective bins of 2˚C soil temperature and 5% APFS were observed at the vegetated set-aside site (SA-veg), for soil temperatures around 20˚C and AFPS between 65 and 75%.

Greenhouse Gas Budget

Based on the flux data obtained in this study, we were unable to compare the full GHG budget of the two sites, since our measurements did not include GPP, C losses from water and discharge. In a comprehensive study on an abandoned peat meadow in the The Netherlands, Hendriks et al. (2007) found the area to be a small C sink after rewetting (-86 g CO_2-eq. m^{-2} yr^{-1}). Although we found significant differences for N_2O and CH_4 between the sites, the fluxes were generally low. Despite higher global warming potential (GWP) compared with CO_2, neither N_2O nor CH_4 is expected to change the picture of the set-aside site as a larger source of respiration than the cultivated site. Based on the results obtained using manual chambers, the contribution of N_2O to GWP at the sites (expressed as CO_2-equivalents; **Table 2**) was 0.27% from the bare set-aside soil and 4.33% from the cultivated site, while for CH_4 the contribution was negative for all plots (-0.14, -0.18, and -0.10% from SA-bare, SA-veg, and the cultivated site, respectively). However, N_2O emissions in particular were influenced by the vegetation cover and could be higher when considering the whole year. Our results therefore support the conclusion reached in previous studies that N_2O and CH_4 are of minor importance for the GHG balance of cultivated peat soils.

CONCLUSION

Based on measurements obtained using manual and automatic chambers, ecosystem respiration from drained peat soil was higher at a set-aside site than at a continuously cultivated site. The cultivated site was a greater source of N_2O and a smaller sink for CH_4 compared with the set-aside site, but this had only a small effect on the GHG balance derived in this study. However, in order to compare the full GHG balance, more continuous measurements are needed and GPP, C losses through water and discharge should be taken into account.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/Supplementary Material, further inquiries can be directed to the corresponding author.

AUTHOR CONTRIBUTIONS

ÖB planned and set up the experiment and collected the data. KM analysed the data and provided a first draft of the manuscript. ÖB and TK reviewed and edited the manuscript.

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