Emissions of CO\textsubscript{2}, N\textsubscript{2}O and CH\textsubscript{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\textsubscript{2}, while fluxes of N\textsubscript{2}O and CH\textsubscript{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\textsubscript{2}, N\textsubscript{2}O, and CH\textsubscript{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\textsubscript{2}O fluxes and lower CH\textsubscript{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\textsubscript{2}) from the atmosphere (Gorham 1991), but at the same time emit substantial quantities of C as both CO\textsubscript{2} and methane (CH\textsubscript{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\textsubscript{2} and nitrous oxide (N\textsubscript{2}O) fluxes, but to increase the emissions of CH\textsubscript{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 erosion (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 CO₂ emissions (e.g., Joosten and Clarke, 2002; Lohila et al., 2004; Joosten 2009; Froling et al., 2011; Salm et al., 2012; Ballantyne et al., 2014). Moreover, these highly elevated CO₂ 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 CO₂-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 gleyta) (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 CO₂ and N₂O emissions (Kasimir-Klemendtsson et al., 1997; Maljanen et al., 2010). Enhanced nutrient availability through fertilizer application and drainage has been shown to increase N₂O emissions (Alm et al., 2007; Teh et al., 2011; van Beek et al., 2011; Salm et al., 2012). Emissions of CH₄ have been found to decrease due to soil erosion and management, suggesting that cultivated peat soils may even act as CH₄ 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, N₂O and CH₄ 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 CO₂ (Kasimir-Klemendtsson et al., 1997; Maljanen et al., 2003; Smith and Conen 2004; Maljanen et al., 2010). 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 CO₂ from the atmosphere than an adjacent undisturbed bog. On including emissions of N₂O and CH₄, Wang et al. (2020) found that the same abandoned pasture had a cooling effect, primarily resulting from changes in CO₂ and CH₄, while the bog acted as nearly climate-neutral. Based on these findings, the authors reiterated the suggestion that abandoning cultivated peatland sites could mitigate CO₂ emissions. In contrast, Maljanen et al. (2007) found that abandoned organic soils in Finland showed slight net CO₂ uptake during the growing season, but were CO₂ sources outside that period, and that mean annual net CO₂ 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 CO₂, N₂O, and CH₄, 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 at the margin of a natural peatland area about 30 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 (N₉₀ₒₒ) and total carbon (C₉₀ₒₒ) 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...
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 CO₂ 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.
(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., 2016a); and 2) together with N$_2$O 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$_2$O 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$_2$O 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$_2$O 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 plant 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$^\circ$ to 30$^\circ$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$^{-2}$), 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 dplyr 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 (N2O) and methane (CH4) 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 | CO2 [mg m−2 h−1] | N2O [µg m−2 h−1] | CH4 [µg m−2 h−1] | GWP[a] [mg m−2 CO2-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 | -9  | -458 | 67  | 36 | 0   | -778 | 237 | 36 |
| 2013      | 401   | 144 | 1,098 | 58 | 7   | -474 | 503 | 129 | -37 | -221 | 67  | 128 |
| 2015      | 1,018 | 224 | 1,999 | 54 | 0   | -296 | 354 | 48 | -56 | -345 | 69  | 48 |
| Average   | 743   | 144 | 2,547 | 160b | 0   | -474 | 503 | 213b | -36 | -778 | 237 | 212b | 742 |
| Bare soil (bare) |     |      |     |     |     |      |     |     |     |     |      |     |     |
| 2012      | 880   | 192 | 4,102 | 54 | 0   | -81  | 85  | 24 | -33 | -166 | 11  | 24 |
| 2015      | 169   | 98  | 561  | 10  | 0   | -81  | 85  | 24 | -33 | -166 | 11  | 24b | 790 |
| Average   | 791   | 98  | 4,102 | 64b | 0   | -81  | 85  | 24b | -33 | -166 | 11  | 24b |
| Cultivated |     |      |     |     |     |      |     |     |     |     |      |     |     |
| 2012      | 560   | 69  | 1,661 | 72 | 0   | -156 | 719 | 36 | 0   | -56  | 545 | 36 |
| 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   | 415   | 43  | 1,661 | 239b | 30  | -207 | 2,447 | 235b | -5 | -373 | 545 | 237b | 424 |

*Median over the study period.
**Total number of data points.
†Calculated using the GWP of N2O (298) and CH4 (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.

| CO2 [mg m−2 h−1] | N2O [µg m−2 h−1] | CH4 [µg m−2 h−1] |
|------------------|--------------------|------------------|
| Site             | (µg m−2)          | (µg m−2)        | (µg m−2)    |
| SA-veg           | A                  | B                | B            |
| SA-bare          | A                  | B                | B            |
| Cultivated       | B                  | A                | A            |

Year             | (µg m−2)          | (µg m−2)        | (µg m−2)    |
| 2012             | A                  | n.s              | a            |
| 2013             | A                  | n.a              | A            |
| 2015             | B                  | A                | B            |
| 2016             | A                  | B                | B            |

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

TABLE 4 | Season-specific average plant-derived carbon dioxide (CO2) flux [mg m−2 h−1] at the set-aside site in seasons covered by manual measurements in 2012 and 2015.

| Season          | Average CO2 flux [mg m−2 h−1] |
|-----------------|------------------------------|
|                 | Vegetated | Bare soil | Plant-derived [%] | n |
| Spring (until 22 June) | 896 b      | 542 ab     | 40 2             | 40 |
| Summer (until 15 August) | 1,118 a | 864 a      | 23 6             | 6  |
| Autumn (until 1 November) | 398 c     | 215 b      | 46 9             | 9  |

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.

Differences between SA-veg and SA-bare were found (Table 3). According to the IPCC (2013) emission factors, CO2 emissions should be much lower from the set-aside site (grassland, drained, boreal: 5.7 t CO2-C ha−1 yr−1) compared with the cultivated site (cropland, drained, boreal: 7.9 t CO2-C ha−1 yr−1), but this was not confirmed by our data.

Plant-derived CO2, estimated as the difference between CO2 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 CO2 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 CO2 fluxes were lower in summer than in spring and autumn (Table 4). This can be explained by the significantly higher CO2 fluxes from both SA-veg and SA-bare during summer (p < 0.01).

Soil temperature and soil erosion status had a strong impact on formation and release of CO2, 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 CO2 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 CO2 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).

N2O
Median fluxes of N2O 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)
showed zero N$_2$O fluxes on average. This was mainly due to zero fluxes during summer and autumn, while fluxes were negative during spring (−39 µg m$^{-2}$ h$^{-1}$; Table 5). In SA-veg, the N$_2$O fluxes were similar in spring and autumn (7 and 5 µg m$^{-2}$ h$^{-1}$, respectively) and were slightly, but not significantly, higher in summer (14 µg m$^{-2}$ h$^{-1}$). At the cultivated site, N$_2$O fluxes were not significantly different during spring and summer (141 and 288 µg m$^{-2}$ h$^{-1}$, respectively), but were significantly lower ($p < 0.05$) in autumn (27 µg m$^{-2}$ h$^{-1}$).

In general, N$_2$O 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$_2$O emissions, due to lack of N uptake by plant roots. However, we found that vegetated plots acted as N$_2$O sources, while fluxes were on average negative for bare soil plots (Table 2). Johansson et al. (2011) found a negative correlation between N$_2$O 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$_2$O as a substitute. Although it can be assumed that denitrification was the main process for N$_2$O production at all sites in this study, the negative fluxes found for SA-bare indicate reduction of N$_2$O 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$_2$O 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$_2$O 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$_2$O 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-Klemetsson 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 µg m$^{-2}$ h$^{-1}$) than at the cultivated site (5 µg m$^{-2}$ h$^{-1}$). Although CH$_4$ uptake by SA-bare (33 µg 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 µg m$^{-2}$ h$^{-1}$) and high uptake rates (778 µg m$^{-2}$ h$^{-1}$). The function of all sites as zero emissions or even a net sink

![FIGURE 2](https://example.com/image.png) Carbon dioxide (CO$_2$) flux [g 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-bare [●], SA-veg [○]) and the cultivated site (cult. ▲). Significant correlations ($p < 0.05$) are shown by the respective regression line.

**TABLE 5** | Season-specific median nitrous oxide (N$_2$O) flux [µg m$^{-2}$ h$^{-1}$] from the set-aside site (SA-bare and SA-veg) and the cultivated site.

| Season                | Median N$_2$O flux [µg 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.
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 (data not shown), but this relationship was not significant at either of the sites. Similarly, Kasimir-Kлемдтsson 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-Kлемдтsson et al. (2009), we did not find a significant correlation with soil temperature or with the

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

![FIGURE 5](image1.png) 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](image2.png) 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.

![FIGURE 7](image3.png) Daily ecosystem respiration (g m$^{-2}$ d$^{-1}$) 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 | Bare soil | Cultivated |
| 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).
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 CO2 followed a seasonal trend, with higher emissions during the summer months and decreasing emissions as winter approached (Figure 6).

In July, CO2 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 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 7](image-url)
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% AFPS 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 CO₂-eq. m⁻² yr⁻¹). Although we found significant differences for N₂O and CH₄ between the sites, the fluxes were generally low. Despite higher global warming potential (GWP) compared with CO₂, neither N₂O nor CH₄ 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 N₂O to GWP at the sites (expressed as CO₂-equivalents; Table 2) was 0.27% from the bare set-aside soil and 4.33% from the cultivated site, while for CH₄ 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, N₂O 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 N₂O and CH₄ 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 N₂O and a smaller sink for CH₄ 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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Conflict of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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