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Effects of fire on CO_2, CH_4, and N_2O exchange in a well-drained Arctic heath ecosystem

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Abstract

Wildfire frequency and expanse in the Arctic have increased in recent years and are projected to increase further with changes in climatic conditions due to warmer and drier summers. Yet, there is a lack of knowledge about the impacts such events may have on the net greenhouse gas (GHG) balances in Arctic ecosystems. We investigated in situ effects of an experimental fire in 2017 on carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) surface fluxes in the most abundant tundra ecosystem in West Greenland in ambient and warmer conditions. Measurements from the growing seasons 2017 to 2019 showed that burnt areas became significant net CO_2 sources for the entire study period, driven by increased ecosystem respiration (ER) immediately after the fire and decreased gross ecosystem production (GEP). Warming by open-top chambers significantly increased both ER and GEP in control, but not in burnt plots. In contrast to CO_2, measurements suggest that the overall sink capacity of atmospheric CH_4, as well as net N_2O emissions, were not affected by fire in the short term, but only immediately after the fire. The minor effects on CH_4 and N_2O, which was surprising given the significantly higher nitrate availability observed in burnt plots. However, the minor effects are aligned with the lack of significant effects of fire on soil moisture and soil temperature. Net uptake and emissions of all three GHG from burnt soils were less temperature-sensitive than in the undisturbed control plots. Overall, this study highlights that wildfires in a typical tundra ecosystem in Greenland may not lead to markedly increased net GHG emissions other than CO_2. Additional investigations are needed to assess the consequences of more severe fires.

KEYWORDS
carbon dioxide, GHG balance, Greenland, methane, NEE, nitrous oxide, tundra

1 | INTRODUCTION

In the Arctic, the air temperature has increased more than twice the global average during the last few decades (Meredith et al., 2019) and may increase by 2–8°C before 2100 (Meredith et al., 2019), amplified by feedback mechanisms such as decreased sea ice extent Johannessen et al. (2016) and changes in surface albedo due to snow cover changes (Chapin et al., 2005). Such changes in climatic conditions (including precipitation) are expected to alter the exchange of methane (CH_4), carbon dioxide (CO_2), and nitrous oxide (N_2O),

between soil and atmosphere in terrestrial ecosystems of the ice-free part of the Arctic (Schuur et al., 2015). These three important greenhouse gasses (GHG) are expected to respond very differently to the same set of climate change (Schuur et al., 2015). A limited number of studies address the GHG budget in the Arctic considering all three gasses (Wagner et al., 2019), of which only a few are linked to shifts in climate conditions other than warming (Voigt et al., 2017) and even fewer studies focus on event-driven changes (Virkkala et al., 2018). This is despite those events such as fires, storms, warm spells, and heavy rain may be as important as more slow and steady changes (Masrur et al., 2018). At high latitudes, the frequency and extension of tundra fires are related to climatic conditions (Hu et al., 2015), and the increased occurrence of fire events that have been linked to changes toward drier and warmer summers (Mack et al., 2011; Masrur et al., 2018). In total, an area of approximately 0.12% of the Arctic tundra has burnt in the time of 2002–2013 (Hu et al., 2015) and models project an increase of up to 200% of the annual burnt area by 2100 in some areas (Young et al., 2017). Although there are large regional differences and tundra fires occurred primarily in Alaska and northeastern Siberia (Hu et al., 2015) a shift in the climate may lead to increased fire frequency in regions with currently low or no fire activity (Chipman et al., 2015; Hu et al., 2015). Fire events can markedly disturb tundra ecosystems in different ways, including above- and belowground plant biomass destruction (Racine et al., 1987) and alter soil properties through changes in the availability of soil phosphorus (P), nitrogen (N), and carbon (C) (Hu et al., 2015). Such changes may subsequently alter soil infiltration, albedo, water content, and the energy balance (Beyers et al., 2005).

Approximately half of the global soil C is deposited in the Arctic due to very slow decomposition rates of soil organic matter (SOM) controlled by prevailing low temperature and wet conditions (Davidson & Janssens, 2006; Hugelius et al., 2014). Thus, climate change and increased fires in these regions can lead to high amounts of C released into the atmosphere, affecting the global C budget, and leading to positive climate feedback (Masrur et al., 2018). However, the magnitude of fire effect on SOM and soil processes depends predominantly on the fire severity (Knicker, 2007; Neary et al., 1999). There is still only a poor understanding of the short- and long-term effects of fire on the C and N cycle in tundra ecosystems (Hu et al., 2015; Masrur et al., 2018). Direct effects of fires include emissions of CO_2 and N_2O due to aboveground biomass combustion (Bret-Harte et al., 2013) as well as a nutrient N flush, increasing the soil mineral N concentrations through pyrolysis (Dannenmann et al., 2018; Kulmala et al., 2014). However, the rise in nutrient concentration in soils following vegetation destruction may additionally be affected by dead roots and other organic material and the absent or reduced plant uptake of nutrients (Kulmala et al., 2014; Rasmussen et al., 2020). Vegetation recovery (and corresponding changes in the soil) after a tundra fire seems to depend on the fire severity (Kelly et al., 2021) as well as the availability and uptake of inorganic N (Jiang et al., 2015). A comprehensive study suggests fire severity at higher latitudes is primarily driven by belowground fuel availability in form of biomass and secondarily by aboveground biomass (Walker et al., 2020).

Studies investigating the long-term effects of fire events on CH_4 gas exchange in boreal forests (4–89 years) found that fires can lead to increased CH_4 uptake rates in well-drained soils, which was linked to decreased soil moisture and increased soil temperature (Song et al., 2017; Takakai et al., 2008). Otherwise, the impacts of fire events on soil GHG fluxes in tundra ecosystems remain unclear. Overall, there seem to be a lack of quantification of immediate and short-term net exchange rates of CO_2, CH_4, and N_2O following an experimental fire in the Arctic.

Here, we report immediate (after 1–27 days) and short-term (after 1 and 2 years) in situ measurements of CH_4, CO_2, and N_2O fluxes in a well-drained tundra ecosystem in West Greenland following experimental fire. The fire impacts are investigated under ambient and warmer climate conditions by deployment of open-top chambers (OTC). We test the following hypotheses as a consequence of fire: (1) an expected immediately shift from a C sink to a net C source, (2) soil CH_4 oxidation capacity is immediately affected but re-establishes over time, and (3) immediate N_2O emissions increase following nutrient release but emissions decrease again on a short-term basis. Our aims were to (1) quantify immediate and short-term net CO_2, CH_4, and N_2O exchange rates following an experimental fire, (2) identify factors that limit processes responsible for net ecosystem GHG exchange (3) test the relationship between temporal changes in soil physicochemical conditions and net GHG exchange, and (4) quantify effects of fire on net GHG exchange rates under climate warming.

2 | METHODS

2.1 | Site description

The study area is in Blæsedalen ("The Windy Valley") at the southern tip of Disko Island (69°16′N, 53°27′W), in West Greenland. Annual and summer mean (±SD) air temperatures (1991–2017) were −3±1.8 and 6.8±1.3°C, respectively, and mean annual precipitation (1991–2017) was 418±131 mm (Zhang et al., 2019). Mean air temperatures suggest discontinuous permafrost in the area, however, at the study site, permafrost has not been accounted for within the top 1.5 m (Blok et al., 2016). The vegetation at the site is typical for a dry Arctic heath, dominated by dwarf shrubs (height < 10 cm) according to D'Imperio et al. (2017), including deciduous shrubs of *Betula nana* L., *Vaccinium uligionosum* L., *Salix glauca* L., and the evergreen *Cassiope tetragona* (L.) D. Don, in addition to a mixture of mosses (*Tomentypnum nitens* and *Aulacomnium turgidum*), lichens (*Cetraria islandica* (L.) Ach. and *Stereocaulon paschale* spp.) and a sparsely occurrence of herbs and graminoids. The young (<10,000 years) mineral soil with basaltic fragments in the study area is covered by a shallow organic horizon (5–10 cm; Nielsen et al., 2017). Hummocks, resulting from freeze and thaw processes, exist in some parts of the area, however, not in the experiment area.

2.2 | Meteorological data

Air temperature and precipitation during the experiment were available from a weather station located within 500 m to the site

(69°15′N, 53°28′W, 97 m a.s.l.) from the Greenland Ecosystem Monitoring Program. The air temperature sensor (Model CS215, Campbell Scientific) was installed at 2.2 m height and the precipitation gauge (Model ARG100, Campbell Scientific) at 0.6 m height.

2.3 | Experimental setup

The experiment was established in July 2017 on a north-facing slope (5.7° inclination) and treatments included intact controls (control no-OTC = CTO [$n = 5$]), and an experimental fire to burn aboveground vegetation and litter (burnt no-OTC = VBO [$n = 5$]). Additionally, control and burnt plots were paired with the deployment of OTCs, which are labeled with an X in their acronym (control OTC = CTX ($n = 5$) and burnt OTC = VBX ($n = 5$)), giving a total of 20 plots (Figure S1). Treatments were applied to 1.2×1.2 m experimental plots replicated in groups of five 7×7 m blocks (Figure S1). The plots were organized randomly within each block. One day prior to the fire, metal collars were deployed in all plots and simultaneously the aboveground shrub was cut down in the plots prone to fire the next day. This was done to align the fire treatment with an additional "shrub-cutting" treatment, which is not discussed further herein, but see Xu et al. (2021) for details. The shrub-cutting was furthermore implemented as a means to ensure more homogeneous fire dispersal in relatively small plots. The experimental fire occurred on July 29, 2017. The fire was started by using a butane weed-burner (Figure S2) and allowed to burn for 7 min per plot. The 7 min were chosen after a test burning in an adjacent area for estimating the approximate duration and soil temperature increase of a natural fire. For a controlled burning, a mist of water was sprayed onto the plots if the fire continued beyond the 7 min to ensure a homogenous treatment application. A more detailed experiment description can be found in Xu et al. (2021).

2.4 | Soil temperature

Soil temperatures during the burning process in 2 and 5 cm depths were measured in two of the five blocks using an RS-PRO 1384 data logging thermometer (RS Components) equipped with four thermocouples, type K. Temperatures were measured and logged at 5 s intervals.

For continued, longer-term soil temperature records thermocouples were inserted into 2 and 5 cm depths and connected to custom-build data loggers recording the temperature at 30 min intervals. Three sets of loggers were installed on July 31st, 2017. Another two sets were added in July and August 2018.

2.5 | Soil analysis

Soil cores of the top 0–5 cm soil (2 cm diameter) were sampled volumetric in August 2017, July 2018, and July 2019 for analysis of total and soluble C, N, and P. Additionally, samples from 2017 were analyzed for pH and C:N ratio. Subsamples of 5 g field moist soil were weighed out for moisture (105°C for 24 h), pH_{H_2O} (1:2.5 w:vol), and water-soluble extractions (1:5w:vol, 1 h shake). Water extracts were filtered through 2.7 μm membrane filters (Whatman® GF/D) and kept frozen until analysis. Dried and finely homogenized soil samples were folded in tin combustion cups (15–25 mg) for determining total C and N (Flash 2000 elemental analyzer; Thermo Scientific). Contents of nitrate (NO_3^--N), ammonium (NH_4^+-N), and phosphate (PO_4^{3-}-P) in extracts were measured using a flow-injection analyzer (Tecator 5000 FIAStar). Dissolved organic carbon (DOC) and dissolved nitrogen (TDN) were measured using an organic carbon and nitrogen analyzer (Shimadzu TOC/TN Analyzer). Dissolved organic nitrogen (DON) was calculated by subtracting inorganic N (NO_3^- and NH_4^+) from TDN.

2.6 | Gas flux determination

One day before the fire, 20×20 cm square stainless steel collars of 10 cm in height were pushed 5 cm into the soil in each plot. A water-filled groove atop the collar enables a gas-tight seal when the flux chamber is applied. Carbon dioxide fluxes were measured three times in 2017 (July–August), five times in 2018 (June–August), and four times in 2019 (July–August; Table S1). Methane fluxes were measured three times in 2017 (July–August), two times in 2018 (August), and four times in 2019 (July–August; Table S1). Fluxes of CH_4 and CO_2 were measured using transparent polycarbonate chambers (19 or 26.5 cm in height). The chambers were connected either to a portable Gas Analyser Picarro G4301 for CH_4/CO_2 (Picarro Inc.) or to an EGM-5 system for CO_2 only (PP System), circulating the air in a closed loop during the enclosure period of 4 (2017) to 5 min (2018/2019). Changes in chamber CO_2 and CH_4 concentrations were analyzed and logged at 1 s sampling frequency. A fan mounted inside the chamber ensured a homogenous air composition in the headspace. Internal chamber temperature was recorded with a Tinytag temperature data logger (TG-4080; Gemini Data Loggers). Each plot was measured twice, first with a transparent chamber to establish the net ecosystem exchange (NEE). After a short break and time to vent the system and return to equilibrium, a second measurement was taken under dark conditions (chamber covered by a blackout cloth) to establish ecosystem respiration (ER) and CH_4 fluxes. A total of 20 experimental plots was measured within 2 days and the order of plots was changed for individual campaigns. Flux calculation of CH_4 (in mg CH_4 m^{-2} h^{-1}), ER, and NEE (in mg CO_2 m^{-2} h^{-1}) were performed in R version 3.4.3 (R Core Team, 2020). Based on a visual assessment we used a linear or an exponential model fit to changes in CH_4 and CO_2 concentrations over time from the package "HMR" (Pedersen, 2020). Gross ecosystem photosynthesis (GEP) was estimated by subtracting ER from NEE rates.

Nitrous oxide fluxes were measured three times in 2017 (August), seven times in 2018 (June to August), and four times in 2019 (July–September; Table S1). Fluxes of N_2O were measured using the same collars, but with the deployment of a white PVC chamber (10 cm in height) for 160 min. The chamber was pierced by a butyl stopper and

headspace gas samples were collected manually with a 12 ml syringe at times 0, 40, 80, 120, and 160 min. When collecting the gas sample, the headspace was first mixed by vigorously pumping the 12 ml syringe twice. Gas samples were stored in 6 ml (2017 and 2019) or 12 ml (2018) pre-evacuated screw cap Exetainers® (Labco Ltd.). The 20 experimental plots were analyzed within the same day for each date in randomized order. Gas samples were analyzed for N_2O concentration by gas chromatography (Agilent 7890A, ©Agilent Technologies). Fluxes of N_2O ($\mu g\,N_2O\,m^{-2}\,h^{-1}$) were calculated by fitting a linear regression to changes in N_2O over time.

Positive flux values indicate a net release of the respective GHG from the ecosystem into the atmosphere, whereas negative net flux values indicate a net uptake of the respective GHG from the ecosystem.

Soil temperature (5 cm depth; HI93503 thermometer, Hanna Instruments) and soil moisture (%vol at 0–7 cm depth; ML2X Theta Probe, Delta-T Devices) were measured manually in triplicates next to the soil collar during chamber deployments.

The contribution per growing season of CH_4 and N_2O fluxes to the total GHG budget was assessed by harmonizing the fluxes from considered campaigns (Table S1) into CO_2-equivalents ($mg\,CO_2$-eq $m^{-2}\,h^{-1}$). For this conversion, we applied the global warming potential (GWP) relative to CO_2 over a 100-year time horizon, which was 34 for CH_4 and 298 for N_2O (Myhre et al., 2013). Subsequently, fluxes were averaged by growing season and treatment and reported as mean ± standard error of the mean (SE). We included only measurements conducted within the same period to avoid biased estimates.

2.7 | Data and statistical analyses

For the immediate (2017) fire effect, we only included no-OTC plots due to limitations in instrument access and time leading to a lack of data for OTC plots in 2017. Additionally, OTCs were mounted only during the days following the experimental fire. Immediate fire treatment effects on variations in GHG fluxes, soil moisture, and temperature, and soluble soil C, N, and P, were analyzed using a mixed-effects ANOVA. Fixed effects were "treatment group" (CT and VB) and "DAF" (days after fire) and their interactions, "block" was set as random effect, and "plot" as subject identifier. For the analysis of short-term fire and warming effects on variations in GHG fluxes, soil moisture and temperature, and soluble soil C, N, and P, we used the observations from 2018 to 2019 in OTC and no-OTC plots. Here, a repeated ANOVA mixed model was used, with "treatment group" (CT and VB), "warming" (no-OTC and OTC), and their interactions as fixed effects, "block" as random effect, "daf" as repeated effect, and "plot" as subject identifier. The analysis was grouped by the two growing seasons (2018 and 2019). One of the replicates among the burnt OTC plots was identified as an outlier and excluded from the analysis (see Material S1). In both models the least squares post hoc tests on the significant terms were made to assess all the relevant pair-wise comparisons of least squares means.

Tuckey adjusted p-values were used for multi-comparison correction. Residual scatter plots were inspected for the evaluation of normality and homogeneity of variance and if necessary data were subsequently transformed to obtain acceptance of normality.

Differences in soil C:N and soil pH (5 days after fire in 2017) in no-OTC plots were tested with a one-way ANOVA with "treatment" (CTO, VBO) as an independent variable. Differences between burnt and un-burnt plots with respect to continuous soil temperature data were tested with a Welsh t-test. Inter-annual differences in manual measured soil temperature and soil moisture in ambient conditions were tested with a one-way ANOVA with "year" (2017, 2018, and 2019) as an independent variable. Potential correlations between GHG fluxes and soil moisture and temperature were assessed by Pearson correlation. Statistical significance is accepted for $p < .05$. The values in the figures are mean ± SE unless otherwise stated. All statistical analyses were performed using SAS Enterprise Guide 7.1 (SAS Institute Inc., 2014). Figures were made using the package "ggplot2" (Wickham, 2016) in R version 3.4.3 (R Core Team, 2020).

3 | RESULTS

3.1 | Meteorological observations

Meteorological data show that weather conditions in the three summers (June to September 2017–2019) varied markedly between years (Figure 1a). The summer in 2018 had the lowest mean air temperature (4.8°C) when compared to summer 2017 (5.4°C) and 2019 (8°C). The timing of the growing season's start differed especially between the years 2018 (mid-May) and 2019 (start-April). While there was no significant difference in precipitation, the lowest total amount of precipitation during summer was recorded in 2019 (135 mm; Figure 1b). In 2019, manual measured ambient soil temperature (5 cm) was significantly ($p < .05$) higher than in the two previous years (Figure 1d), while soil moisture (0–7 cm) was significantly lower in 2019 compared to the other 2 years (Figure 1c).

3.2 | Surface and soil temperature during the experimental fire

Temperatures during the experimental fire on July 29, 2017 in the shrub and litter layer were above 100°C for in total ca. 4 and 3 min, respectively (Figure 2a). In the top 2 cm of the soil between 40 and 60°C and stayed above 30°C for 5–20 min (Figure 2b). Soil temperatures in 3–5 cm depth did not rise above 15°C during the burning process (Figure 2b). Aboveground and soil temperatures were close to pre-burning temperatures or at least below 30°C within 20 min after the experimental fire (Figure 2). The experimental fire resulted in nearly complete combustion of all aboveground vegetation and scorched stems and litter layer (Figure S4). The continuous soil temperature data indicated no significant differences between unburnt and burnt plots in the three summer periods (2017–2019,
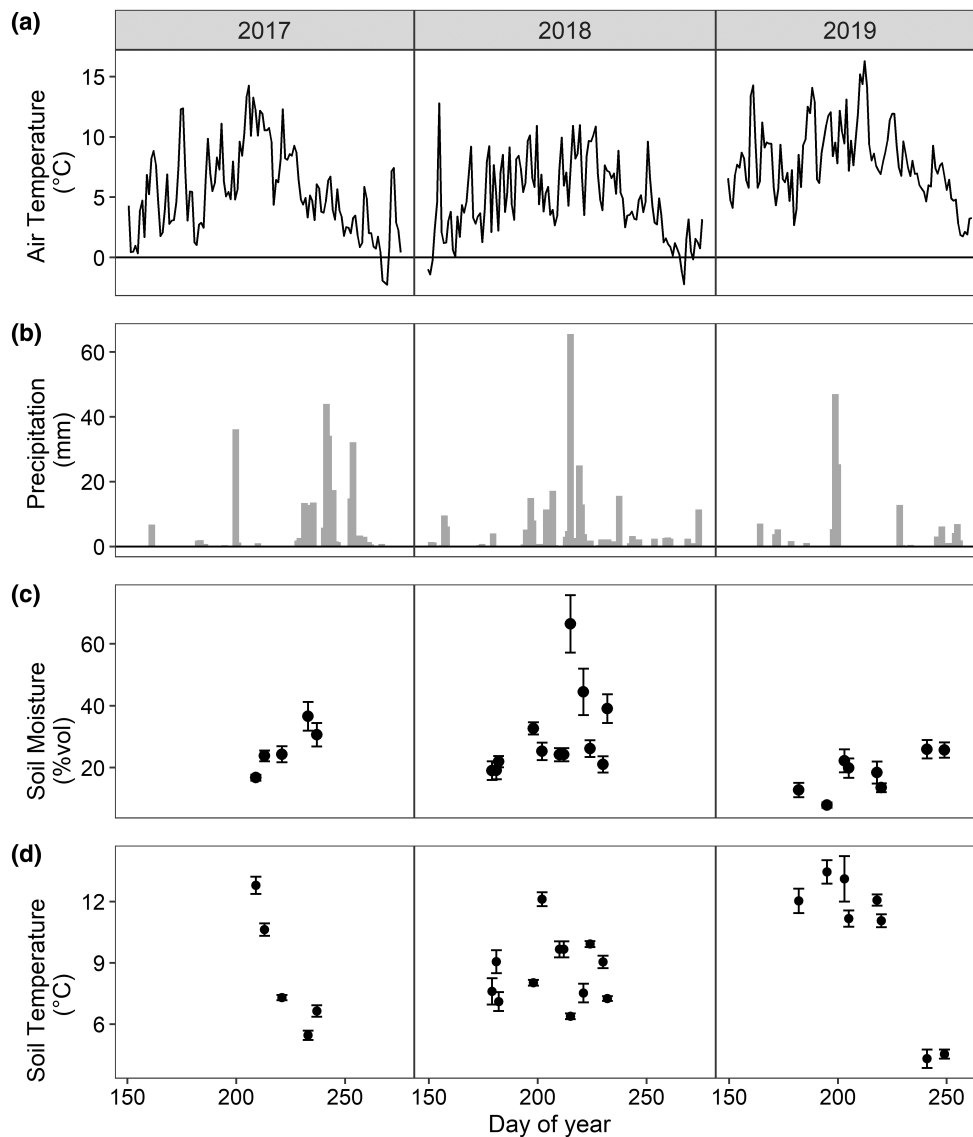
FIGURE 1 (a) Continuous measurements of daily mean air temperature in 2 m height (b) daily sum liquid precipitation from June to September 2017, 2018, and 2019 (only data until September 16, 2019). (b) Manual soil moisture measurements in 0–7 cm depth in control plots, measured during GHG flux measurements. (c) Manual soil temperature measurements in 5 cm depth in control plots, measured during GHG flux measurements. Soil moisture and soil temperature are averaged by campaign ($n = 5 \pm$ SE), measured in Blæsedalen, West Greenland

$p \geq .1$, Welsh t-test) in both depths (Figure S3). However, ambient soil temperatures were significantly higher ($p \leq .05$, Welsh t-test) in 2019 than in 2018 (Figure S3).

3.3 | Soil characteristics and soil water extracts

Soils across all treatment groups were slightly acidic in the top 5 cm measured on DAF5 (Table 1). Soil C:N ratios in the same soil depth ranged between 18.4 ± 0.6 and 18.9 ± 0.7. There were no significant differences in pH or C:N between treatments (Table 1).

There was only an overall significant immediate fire effect on NO_3^--N ($p = .044$) when taking both DAF5 and DAF21 into account. The analysis of only DAF5 soil samples showed significantly higher

NO_3^--N ($p = .01$) and NH_4^+-N ($p = .02$) in burnt plots than in controls (Figure 3a,b; Table S2). This significant difference was not detected in samples taken on DAF21 (Figure 3a,b; Table S2). The same pattern was seen in PO_4^{3-}-P (Figure 3c), DON (Figure 3d), and DOC (Figure 3e) concentrations, but without any significance regarding treatments. The concentration of NO_3^--N was low in burnt and control plots in samples from 2018 to 2019 (Figure 3a). There was no overall significant short-term fire effect taking no-OTC and OTC plots into account. However, in 2018 burnt no-OTC plots (VBO) had significantly higher concentrations of NO_3^--N, NH_4^+-N, DON, and DOC ($p \leq .05$; Figure 3; Table S2) than control no-OTC plots (CTO). There were no significant fire effects in 2019 no-OTC plots, as well as in OTC plots 2018/19. The only observed significant warming effect was in 2018, with significantly ($p = .023$)

(a)

(b)
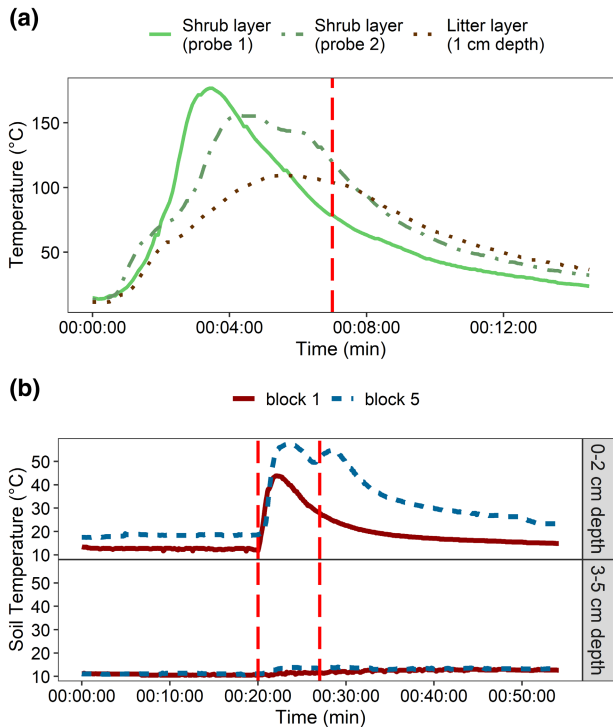
FIGURE 2 (a) Surface temperature during the test burning process as part of the tundra fire experiment on July 29, 2017 in Blæsedalen, Disko Island West Greenland. The red vertical line marks the natural end of the test burning after 7 min (b) Soil temperature in 0–2 and 3–5 cm depth during the burning process in two burning treatment plots (block 1 and 5) in Blæsedalen, Disko Island West Greenland. The vertical red lines mark the start (left) and end (right) of the 7 min experimental fire treatment

higher DON concentrations in control OTC plots (CTX) than in control no-OTC plots (CTO; Table S2).

3.4 | Fire effects on CO_2

Soil moisture and soil temperature measured during the GHG flux measurements were not significantly affected by fire (Figure 5a,b) but followed the same pattern as control plots, which is aligned with the variations in weather conditions observed during the study period (Figure 5a,b; Tables S3 and S4).

Burning of vegetation significantly ($p<.05$; Tables S3 and S4) increased net CO_2 loss reflected by enhanced NEE rates when compared to controls over the entire study period (Figure 4a). The stimulating effect of fire on CO_2 loss in no-OTC was immediate and short-term as indicated by the significant ($p = .001$) increase in NEE when compared to controls (NEE DAF3: CTO: 89 ± 27 mg CO_2 m^{-2} h^{-1} vs. VBO: 266 ± 44 mg CO_2 m^{-2} h^{-1}), and lasted for 1 year ($p = .006$) after the fire. Only in 2019, NEE rates in no-OTC control (CTO) and no-OTC burnt (VBO) plots did not differ significantly. All NEE rates in 2017 were consistently positive (Figure 4a), showing that the study site was a net CO_2 source regardless of treatments. The highest positive NEE rates were measured 1 day after the metal collar

installation and 1 day before the experimental fire. Burnt plots (VB) remained a net CO_2 source in growing seasons 2018/2019, indicated by persisting positive NEE rates, while control plots (CT) were net CO_2 sinks in the same period.

GEP rates in burnt plots (VB) were significantly lower (Tables S3 and S4; less negative) than in controls (CT; Figure 4b) consistent with reduced photosynthetic activity following the fire (Table 1). Immediately after the fire, this was significant ($p = .014$) (Figure 4b; Table S3). Results from the short-term effect analysis showed a persistent significant ($p = .014$) difference between control (CTO) and burnt (VBO) plots in the growing season 2018 (Table S4). GEP rates in burnt plots (VBO) were still affected by the fire in 2019, yet they were more negative suggesting increased photosynthetic activity with no significant difference to control plots (CTO; Table S4).

There was an overall significant fire treatment effect over the entire study period in ER fluxes (Figure 4c; Tables S3 and S4) taking both no-OTC and OTC plots into account. Immediately after the fire no-OTC burnt plots (VBO) had significantly higher ($p = .019$) ER rates than controls (Figure 4c). In contrast, there was no short-term effect of the fire treatment in no-OTC plots, and ER rates did not differ between burnt (VBO) and controls (CTO) 1 and 2 years after the experimental fire. The highest fluxes of ER in control plots were measured the day before the fire, which was also the day after the metal collar installation (ER CTO: 334 ± 59 mg CO_2 m^{-2} h^{-1}).

3.5 | Fire effects on CH_4

All CH_4 fluxes were negative, indicating that the soils were consistently a CH_4 sink throughout the measuring period in all three sampling years in both control and burnt plots (Figure 4d). There were no significant differences in net CH_4 uptake rates between burnt (VBO) and control plots neither immediately after nor 1 or 2 years after the experimental fire. However, net CH_4 uptake in burnt plots was significantly lower ($p = .009$) on DAF3 than on DAF27 (Table S3). Overall, the lowest net CH_4 uptake rates were measured in 2018, while in 2017 and 2019 net CH_4 fluxes were in similar ranges, though this was a non-significant trend.

3.6 | Combined fire and summer warming effects on CO_2 and CH_4 fluxes

When taking all measurements from 2018 to 2019 into account, summer warming by OTCs significantly increased the 2018 and 2019 mean soil temperature in 5 cm depth (Table S4) in all treatment groups (CT/VB), in controls by 0.9°C and in burnt by 1.3°C. However, the effect was only evident in 2018 when looking at single groups, and mainly in control plots (Figure 5b). There was no effect of warming by OTCs on soil moisture in any of the treatment groups or years (Figure 5a; Table S4).

Summer warming by OTC had no significant effect on NEE in neither the burnt group (VB) nor the control group (CT; Figure 4a;

TABLE 1 Seasonal averages (mean (SE)) of measured CO_2, CH_4, and N_2O fluxes, and main soil characteristics by treatment and year, Disko Island, West Greenland

	No OTC warming		OTC warming	
	CTO	VBO	CTX	VBX
2017				
NEE ($mg\,CO_2\,m^{-2}\,h^{-1}$)	98.24 (15.6)	264.36 (24.69)		
GEP ($mg\,CO_2\,m^{-2}\,h^{-1}$)	−79.77 (13.2)	−18.66 (7.49)		
ER ($mg\,CO_2\,m^{-2}\,h^{-1}$)	178.01 (16.69)	272.04 (19.57)		
CH_4 ($mg\,CH_4\,m^{-2}\,h^{-1}$)	−0.07 (0.01)	−0.06 (0.01)		
N_2O ($\mu g\,N_2O\,m^{-2}\,h^{-1}$)	0.45 (1.0)	−1.3 (0.2)		
Soil moisture (%vol)	28.4 (1.89)	30.63 (1.53)		
Soil temperature (°C)	7.61 (0.46)	8.1 (0.48)		
Litter layer C (%)	35.6 (2.6)	30.5 (5.7)		
Litter layer N (%)	0.89 (0.11)	0.84 (0.14)		
Litter layer C:N	41.6 (4.5)	35.8 (1.8)		
Top soil C (%)	7.3 (1.2)	10.5 (1.9)		
Top soil N (%)	0.39 (0.07)	0.57 (0.1)		
Top soil C:N	18.7 (0.9)	18.4 (0.6)		
Top soil pH	6.1 (0.9)	6 (0.6)		
2018				
NEE ($mg\,CO_2\,m^{-2}\,h^{-1}$)	−0.4 (14.1)	107.09 (7.87)	−48.86 (28.22)	98.93 (13.84)
GEP ($mg\,CO_2\,m^{-2}\,h^{-1}$)	−120.53 (14.63)	6.8 (8.28)	−267.83 (11.71)	−27.48 (13.03)
ER ($mg\,CO_2\,m^{-2}\,h^{-1}$)	120.14 (7.36)	100.29 (9.05)	218.98 (11.71)	126.42 (9.97)
CH_4 ($mg\,CH_4\,m^{-2}\,h^{-1}$)	−0.03 (0.01)	−0.04 (0.01)	−0.07 (0.02)	−0.04 (0.01)
N_2O ($\mu g\,N_2O\,m^{-2}\,h^{-1}$)	1.21 (1.2)	1.87 (1.4)	0.95 (1.1)	1.68 (1.3)
Soil moisture (%vol)	30.27 (2.06)	30.74 (2.33)	29.6 (2.51)	33.61 (2.51)
Soil temperature (°C)	8.62 (0.22)	8.88 (0.21)	9.33 (0.26)	9.89 (0.28)
2019				
NEE ($mg\,CO_2\,m^{-2}\,h^{-1}$)	−57.32 (23.83)	85.44 (20.45)	−112.03 (48.88)	73.99 (16.71)
GEP ($mg\,CO_2\,m^{-2}\,h^{-1}$)	−194.86 (36.04)	−33.43 (21.58)	−427.28 (71.19)	−91.37 (20.41)
ER ($mg\,CO_2\,m^{-2}\,h^{-1}$)	137.54 (16.67)	115.33 (14.07)	315.25 (32.95)	165.36 (15.88)
CH_4 ($mg\,CH_4\,m^{-2}\,h^{-1}$)	−0.07 (0.01)	−0.07 (0.01)	−0.11 (0.01)	−0.09 (0.01)
N_2O ($\mu g\,N_2O\,m^{-2}\,h^{-1}$)	−1.61 (1.5)	−1.58 (1.9)	−2.35 (1.9)	−1.52 (1.7)
Soil moisture (%vol)	18.24 (1.31)	18.63 (1.3)	16.66 (1.54)	19.49 (1.34)
Soil temperature (°C)	10.05 (0.59)	10.48 (0.64)	11.21 (0.62)	11.86 (0.69)

Table S4). All burnt plots; with and without warming, were net CO_2 sources.

There was no significant warming effect on GEP rates in burnt plots (VBO/VBX) while warming significantly increased ($p < .05$) GEP rates in control OTC (CTX) when compared to control no-OTC plots (CTO; Figure 4b; Table S4).

In the control group (CT), warming by OTCs significantly ($p < .05$; Figure 4c; Table S4) stimulated ER in all campaigns in the growing seasons 2018/19. However, the effect of OTC warming on ER was not observed in burnt plots (VB), where OTC warming did not significantly increase ER.

Net CH_4 uptake in control plots was significantly ($p < .022$; Table S4) increased by OTC warming in 2018, but not in 2019

(Figure 4d; Table S4). In burnt plots (VB) warming slightly increased net CH_4 uptake across all campaigns, although not significantly.

3.7 | Fire effects in warming plots

Fire effects in OTC plots were generally similar to the results in no-OTC plots (Table S4) with no effects on soil moisture, soil temperature, and net CH_4 uptake rates. Gross ecosystem production was significantly lower in burnt OTC plots (VBX) than in control OTC plots (CTX; Table S4). There was a significant difference in positive NEE rates in burnt OTC plots (VBX) when compared to more negative NEE rates in control OTC plots (CTX; Table S4). In the growing
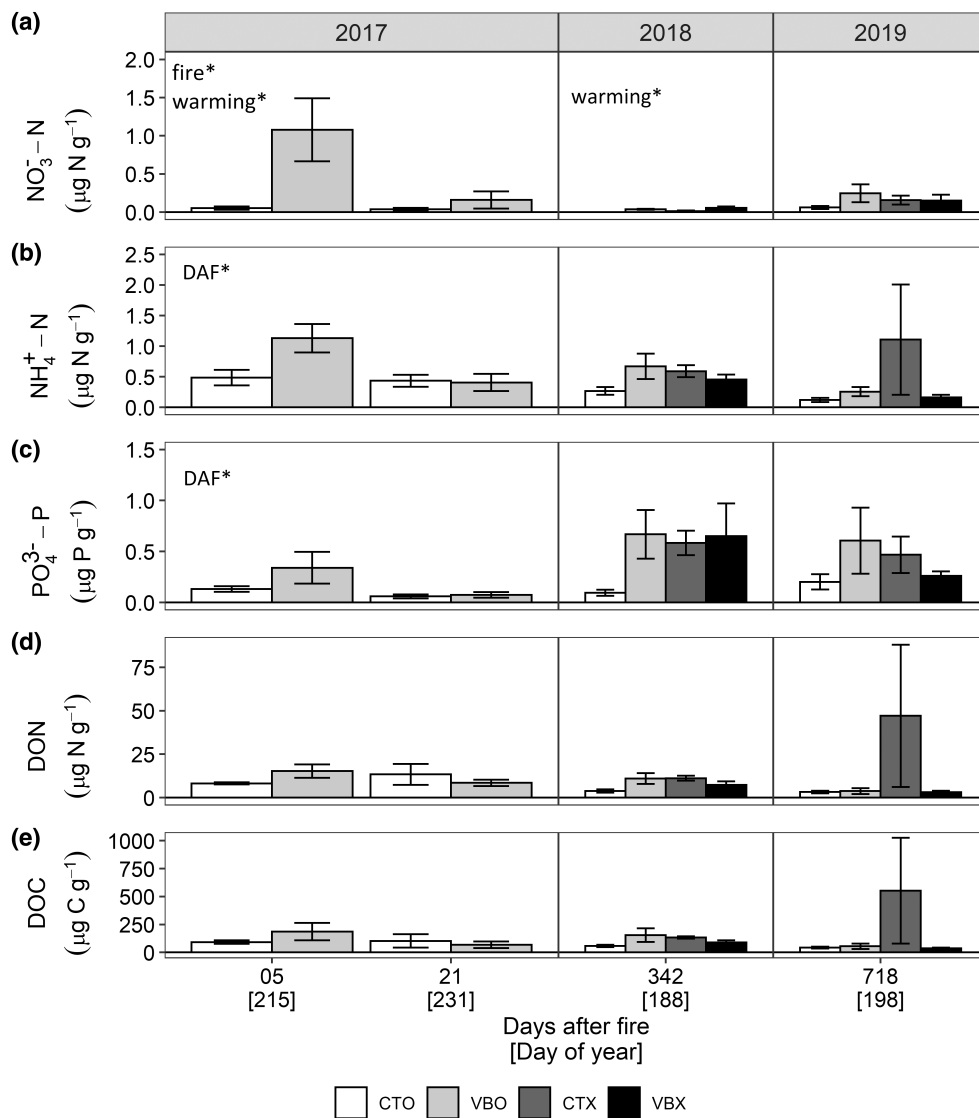
FIGURE 3 Concentrations soluble (a) NO_3^--N, (b) NH_4^+-N, (c) $PO_4^{3-}-P$, (d) DON, and (e) DOC averaged by sampling campaign and treatment ($n = 5 \pm SE$). Overall (no treatment group specific) significant immediate (DAF and fire) and short-term (fire and warming) effects are indicated by asterisk (significance level $p < .05$). For detailed pairwise fixed effect results and p-values see Table S2

seasons 2018/19 burnt OTC plots (VBX) had significantly lower ER rates than control OTC plots (CTX; Table S4).

3.8 | Linkages between CO_2 and CH_4 fluxes, and soil temperature and moisture

Immediately after the fire, CO_2 exchange rates did not correlate with soil moisture or soil temperature in any of the treatments (Figure S6–S8). In the following growing seasons (2018/2019), GEP increased with increasing soil temperatures only in control (CT) no-OTC and OTC plots (Figure S6b). ER increased with increasing soil temperatures in burnt and control plots (VB/CT) (Figure 6b). Soil moisture did not influence CO_2 exchange rates (Figures S6–S8a).

While net CH_4 uptake rates during the growing season 2017 did not correlate with soil moisture in any of the treatments (Figure S9a), the net CH_4 uptake decreased significantly with increasing soil moisture levels in all treatments in the growing seasons 2018 and 2019 (Figure 6a). There was no significant correlation between soil temperatures and net CH_4 uptake (Figure S9b).

3.9 | Fire and warming effects on N_2O fluxes

Surface fluxes of N_2O across all treatments and years were relatively small and in the range of -3 to 2 $\mu g N_2O m^{-2} h^{-1}$ (Figure 4e; Table 1). Immediately after the experimental fire, N_2O fluxes were significantly lower in burnt (VBO) than in control plots (CTO; Figure 4e).
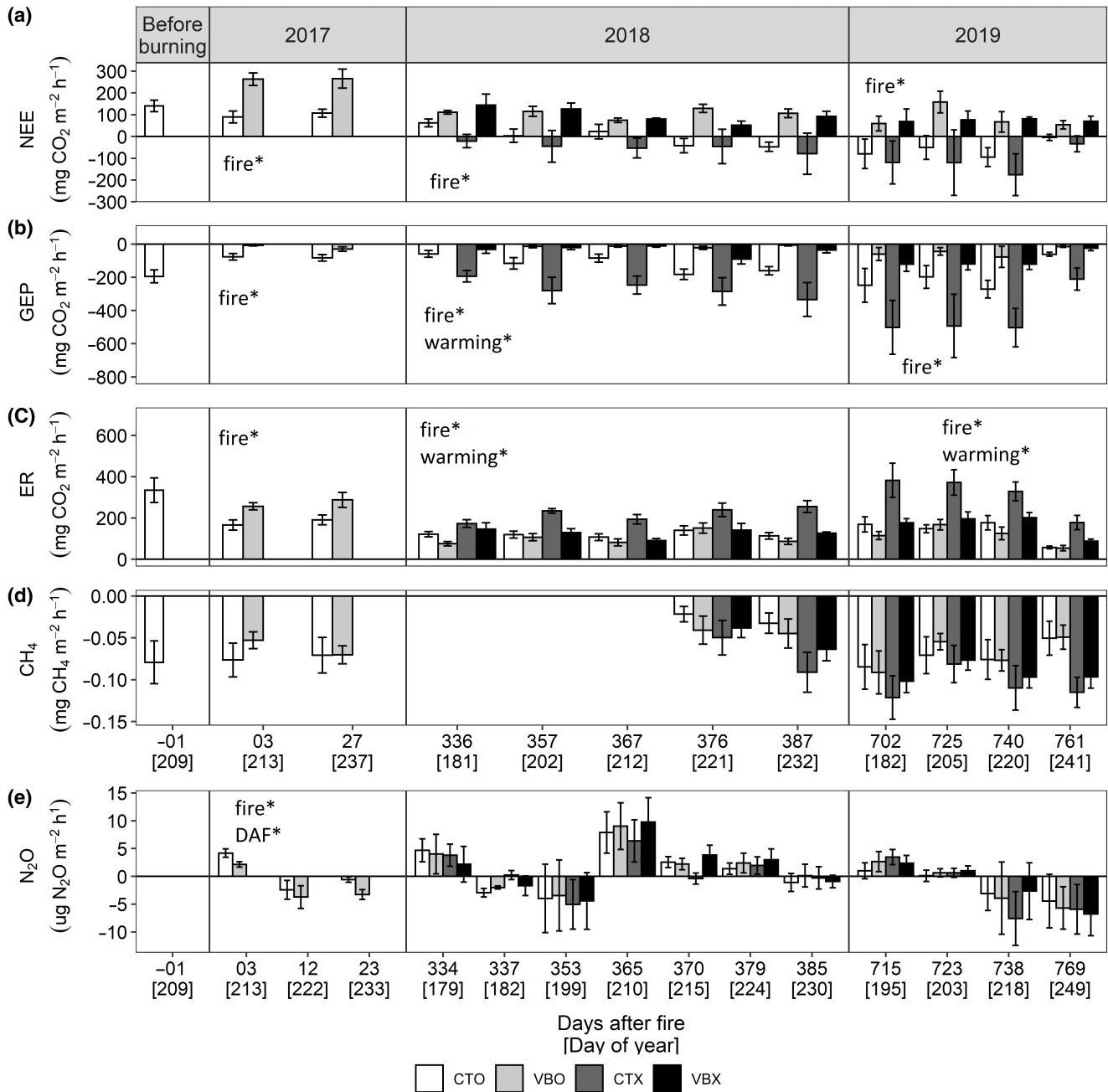
FIGURE 4 Manual measurements of gas fluxes in treatment plots averaged by campaign across the sampling seasons (a) NEE, (b) GEP, (d) ER, (d) CH_4, and (e) N_2O. Overall (no treatment group specific) significant immediate (DAF and fire) and short-term (fire and warming) effects are indicated by asterisk (significance level $p < .05$). For detailed pairwise fixed effect results and p-values see Tables S3 and S4

However, this effect was not significant in the single campaigns 2017 (Figure 4e; Table S3). In the growing season 2017, N_2O fluxes in control plots and burnt plots correlated significantly, positively with manual measured soil temperature in 5 cm depth, leading to less net N_2O uptake and more N_2O release with increasing soil temperatures (Figure S10a). There were no short-term effects of the experimental fire, meaning N_2O fluxes in burnt soils (VBO/VBX) did not differ from control plots (CTO/CTX) in the growing seasons 2018/19. Additionally, we observed no warming effect on N_2O fluxes in any of the treatment groups (Table S4) and they correlated with neither

manual measured soil moisture nor soil temperature in 5 cm depth in the growing seasons 2018/19 (Figure S10).

3.10 | CO_2-equivalents

Estimations of CO_2-eq showed a negligible contribution of CH_4 and N_2O fluxes to the ecosystem GHG budget in terms of global warming potential, observed in both control and burnt areas during the study period (Table 2). While CO_2 fluxes are ranging between −148
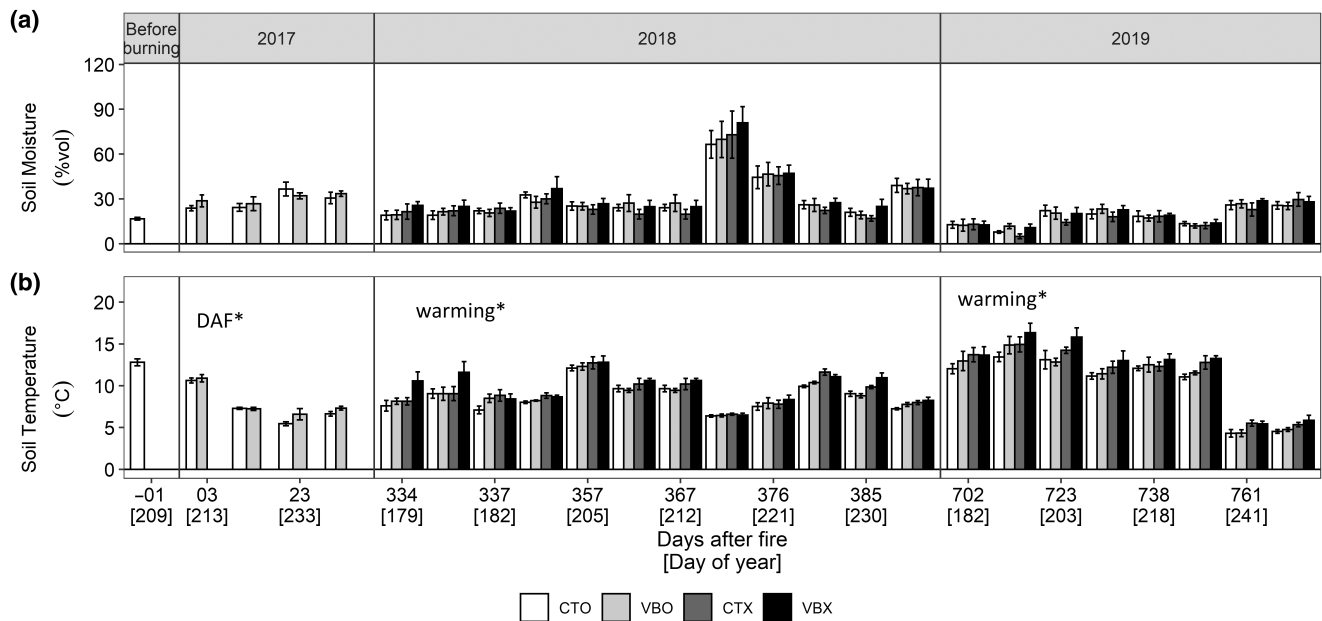
FIGURE 5 Manual measurements of (a) soil moisture in 0–7 cm depth, (b) soil temperature in 5 cm depth averaged by campaign and treatment across the sampling seasons ($n = 5 \pm SE$). Overall (no treatment group specific) significant immediate (DAF and fire) and short-term (fire and warming) effects are indicated by asterisk (significance level $p < .05$). For detailed pairwise fixed effect results and p-values see Tables S3 and S4

and 350 mg CO_2 m^{-2} h^{-1} are CH_4 and N_2O ranges in the lower one-digit area.

4 | DISCUSSION

The goal of this study was to investigate the multiyear effects of experimental fire on CO_2, CH_4, and N_2O fluxes in a well-drained Arctic tundra ecosystem under ambient as well as warmed conditions. Our fire experiment was meant to represent a so-called "ground fire" typically wind-driven, which is the kind of fire most likely to occur in the study area due to the thin organic layer and thus, limited fuel availability (Walker et al., 2020).

4.1 | Fire effects on belowground thermal and moisture regime

The spread and duration of fire depend on many factors, such as wind speed, fuel availability, and moisture, making it a challenge for experimental fires to simulate wildfires realistically (Parra et al., 2012). During a high intensity monitored fire (max flame temperature > 700°C) in a Mediterranean shrubland (organic soil layer <15 cm) an area of 9 ha was burnt within 2.5 h, and despite the extremely high flame temperatures, the topsoil temperatures (0–2.5 cm) in most areas stayed below 100°C during the fire (Stoof et al., 2013). This is in line with observations from other fast-moving fires in light fuel areas like grasslands, with very limited downward heat transfer (Beyers et al., 2005; Knicker, 2007) and our test fire at

the study site in Greenland (Figure 2). This indicates a fire duration time of approximately 7 min under the prevailing weather conditions at the time of experiment start could be comparable to wildfires. Wildfires in dry tundra ecosystems are predominant of moderate or low intensity due to fuel limitations in form of low shrubs and thin organic layers (Maslov et al., 2018; Walker et al., 2020). However, wildfires, such as in the Alaskan Anakutuvuk River area, did not burn homogenously but left areas with different degress of burn severity (moderate and severe; Rocha & Shaver, 2011). Manual soil temperatures in 5 cm depth at our study site (Figure 5b), as well as continuous measurements in 0–2 and 3–5 cm depth (Figure S3), respectively, did not show any lasting effects of fire on soil temperatures, despite a darker surface and corresponding lower albedo. The lack of a significant increase in soil temperature in the topsoil after the fire is in contrast to other studies of tundra wildfires, where the decrease in albedo and the combustion of the insolating moss and organic layer resulted in increased soil temperature and lead to permafrost degradation and increased active layer depth (Hu et al., 2015; Liljedahl et al., 2007; Rocha & Shaver, 2011). However, this effect was most pronounced in largely, severely burnt areas, where up to 15 cm of the pre-fire 18 cm thick organic layer was consumed by fire (Liljedahl et al., 2007). The moss and organic layer at our study site were thin (5–10 cm) and not completely combusted. The albedo change in the small-scaled burnt plots at the north-facing slope has not been sufficient to significantly increase soil temperature. We did not find any difference in soil moisture levels in burnt plots compared to control plots, although we expected an increase after the combustion of the aboveground plants and therefore reduced evapotranspiration (Liljedahl et al., 2007).
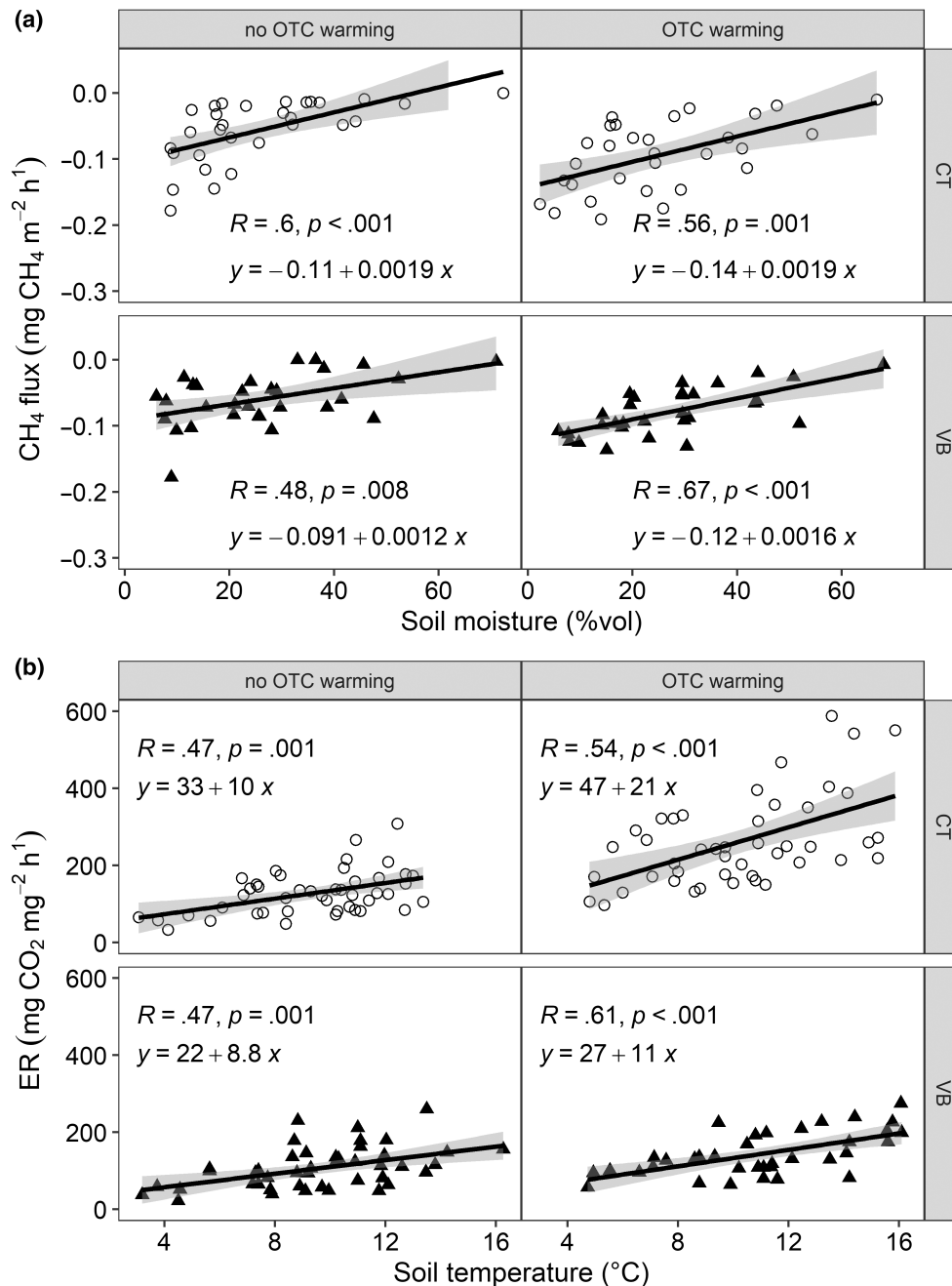
FIGURE 6 Relationship between in situ soil CH_4 uptake rates and soil moisture content (a) and in situ ER rates and soil temperature (b). The measurements were conducted in all treatment plots during the growing season 2018 and 2019. The black solid line indicates the regression line and the gray shaded area indicates the 95% confidence interval. The outlier in the VB OTC group was excluded from the ER correlation

4.2 | Fire effects on soil characteristics and processes

Arctic well-drained soils are generally limited in N and P availability (Street et al., 2018), especially during growing seasons (Rasmussen et al., 2020). Immediately after the fire concentrations of NH_4^+-N and NO_3^--N in burnt plots were significantly higher compared to controls (Figure 3a,b). The same trend of increasing nutrient concentrations immediately after the fire was observed in other studies and

attributed to lower plant uptake, the direct formation of NH_4^+-N during the burning process and the transformation to NO_3^--N in the absence of plant uptake (Dannenmann et al., 2018; Kulmala et al., 2014; Ludwig et al., 2018). In this study, 3 weeks after the fire, there were no differences in nutrient concentrations between burnt and control plots, which was similar to observations from a fire experiment in a boreal forest (Ludwig et al., 2018) and could be due to multiple reasons, such as leaching, infiltration, uptake, or sorption (Knicker, 2007; Ludwig et al., 2018). Significant higher concentrations of N nutrients

TABLE 2 Estimates of seasonal averages (mean (SE)) of CO_2-eq of ER, NEE, CH_4, and N_2O fluxes in control and burnt plots based on ($n = 2$) campaigns per growing season and GHG measured in Blæsedalen, West Greenland

	No OTC warming		OTC warming	
	Control CO_2-eq (mg CO_2 m^{-2} h^{-2})	Burnt CO_2-eq (mg CO_2 m^{-2} h^{-2})	Control CO_2-eq (mg CO_2 m^{-2} h^{-2})	Burnt CO_2-eq (mg CO_2 m^{-2} h^{-2})
2017				
NEE	98.24 (15.6)	264.36 (24.7)		
ER	178.01 (16.7)	272.04 (24.7)		
CH_4	−2.5 (0.5)	−2.09 (0.3)		
N_2O	0.61 (0.3)	−0.17 (0.3)		
2018				
NEE	−44.8 (18.3)	61.25 (35.7)	−62.46 (58.3)	72.05 (15.8)
ER	126.74 (13.6)	144.01 (35.7)	247.17 (20.9)	133.61 (15.4)
CH_4	−0.92 (0.2)	−1.63 (0.4)	−2.39 (0.6)	−1.73 (0.3)
N_2O	0.04 (0.3)	0.38 (0.4)	0.25 (0.4)	0.31 (0.4)
2019				
NEE	−72.62 (33.6)	117.76 (36.2)	−147.92 (84.5)	78.6 (19.3)
ER	162.14 (19.4)	146.37 (20)	350.3 (36.7)	198.55 (19.4)
CH_4	−2.49 (0.5)	−2.23 (0.3)	−3.25 (0.6)	−2.95 (0.3)
N_2O	−0.44 (0.5)	−0.49 (0.9)	−1.03 (0.8)	−0.25 (0.8)

in burnt plots were observed in the growing seasons 2018 (non-significant in 2019), which could be from ash residues and continued low plant uptake (Jiang et al., 2015). There were no effects on soil C:N and pH (Table 1). Because soil chemistry is influenced by many factors, such as organic matter input and environmental factors, our results are only a snapshot of the soil conditions at the time of sampling. When it comes to microbial-driven soil processes, temperatures reached during the experimental fire at the surface may have been high enough to inhibit microbial growth close to the surface (Bárcenas-Moreno & Bååth, 2009; Barreiro et al., 2020). However, microbial growth recovers rapidly, in response to increased nutrients availability and C from dead organisms after fire (Bárcenas-Moreno & Bååth, 2009; Barreiro et al., 2020). The impacts of fire on microbial growth and biomass are determined by the residence time of high temperatures rather than the maximum temperature (Lombao et al., 2020). Therefore, it is likely that the direct effects of the experimental fire on soil microorganisms in this study have been minor due to the limited temperature increase in the soil (Figure 2).

4.3 | Immediate and short-term effects of the experimental fire on CO_2 fluxes

The experimental fire in a dry heath ecosystem that has previously been shown to be a net CO_2 sink (Ravn et al., 2020) turned the burned areas into a net C source. Fire turning burnt areas into larger CO_2 sources due to decreased photosynthesis rates as the main controlling factor was also observed in another tundra fire study (Rocha & Shaver, 2011). Although the here observed NEE rates were significantly higher in burnt plots than in controls, as expected,

controls were also a net CO_2 source in 2017 (Figure 4a; Table 1). This is assumed to depend on the timing of the measurements shortly after the collar installation and is not expected to have biased the relative results of the treatment effect since all plots were uniformly affected.

The significantly increased ER rates in burnt plots immediately after the fire (Figure 4c) may be related to an increase in microbial decomposition due to increased organic input from dead roots or a flush of nutrients due to decreased plant uptakes and charred organic materials in the burnt plots (Kulmala et al., 2014). Since Arctic soils are nutrient limited that could have enhanced microbial activity due to the higher availability of degradable material and residuals in ash for C decomposition as well as nutrients (Jiang et al., 2015), which may offset the assumed decrease in autotrophic respiration due to root damage (Morishita et al., 2015; Rocha & Shaver, 2011). These results confirm our first hypothesis of an immediate shift from a C sink to a net C source. This is in contrast to a study from a tundra shrub removal field experiment that found decreased ER immediately after removal, which was explained by reduced plant root respiration (Ravn et al., 2020).

Measurements in the growing seasons 1 and 2 years after the experimental fire show that burnt plots remained a net CO_2 source (Figure 4a; Table 1). Thus, the remaining low photosynthetic activity in burnt plots (Figure 4b) of the slowly recovering vegetation was not sufficient to balance the ecosystem's CO_2 loss, similar to other fire-influenced tundra ecosystems (Rocha & Shaver, 2011). The time needed for vegetation to recover following fire is of major importance. Especially taking into account the dominant contribution of CO_2 in the ecosystem's GHG budget (CO_2-eq; Table 2) and the very short growing seasons in the Arctic.

In the burnt plots at our study site was a non-significant trend in increasing GEP over time toward the end of the growing season 2018 and in the growing season 2019 (Figure 4b). Bret-Harte et al. (2013) observed a recovery in net primary production of vascular plants within 4 years after a wildfire and Jiang et al. (2015) reported CO_2 sink capacity higher in burnt than in unburnt areas 4 and 5 years after a fire. In this study, we observed a rapid recovery of *Salix glauca* re-sprouting 1 year after the fire (Figure S4). This supports the assumption that, despite the fire, the belowground biomass remained fairly intact and the presence of rhizomes allowed the re-sprout of *Salix* species (Haeussler & Coates, 1986). Bret-Harte et al. (2013) observed a significant decrease in living root biomass in severely burnt areas while living root biomass in moderately burnt areas was comparable to unburnt areas, after a tundra wildfire with an average organic layer loss of 6 cm (Mack et al., 2011). Therefore, wildfires may amplify the expansion of shrubs and thereby alter the plant species composition in the Arctic (Chen et al., 2021). Unburnt plots on the other hand turned into net CO_2 sinks in the growing seasons 2018 and 2019 due to higher GEP rates than immediately after the experiment start (Figure 4a,b; Table 1).

The difference in ER between burnt and control in no-OTC plots was not significant, as also reported in other studies carried out in tundra and boreal forests (Rocha & Shaver, 2011; Ueyama et al., 2013). This is in contrast to a few studies reporting an increase in CO_2 emissions with time after a fire (1 to >100 years after fire; Makita et al., 2016; Ribeiro-Kumara et al., 2020). The lack of increased ER 1 and 2 years after the fire may also be a net result of reduced autotrophic respiration (Morishita et al., 2015) and increased microbial respiration.

4.4 | Immediate and short-term effects of the experimental fire on CH_4 fluxes

All plots were net CH_4 sinks, irrespective of treatments (Figure 4d; Table 1), and CH_4 uptake rates were within the range reported in previous in situ studies from the same study area (D'Imperio et al., 2017; St Pierre et al., 2019). Immediately after the fire, CH_4 uptake was lowest in burnt plots, although not significantly when compared to controls. Previous studies reported that enhanced NH_4^+-N and NO_3^--N concentrations have an inhibiting effect on soil CH_4 oxidation (Fender et al., 2012; Priemé & Christensen, 1997). Thus, fire may have an indirect effect on CH_4 uptake as inorganic N in soil solution can increases after fires (Dannenmann et al., 2018; Kulmala et al., 2014; Ludwig et al., 2018), as observed in this study. Although temperatures near and above 100°C have previously been shown to reduce the overall soil CH_4 oxidation capacity (Ho et al., 2016; Mohanty et al., 2007). The highest potential for soil CH_4 oxidation in Arctic dry soils has been measured in the top 15 cm of the soil, with a maximum at around 10 cm depth (Jorgensen et al., 2015). Therefore, the lack of heat transfer from the surface fire into the soil in this study could explain the lack of significant fire effect on CH_4 uptake.

In the growing seasons 2018/19, net soil CH_4 uptake rates did not differ significantly between burnt and control plots ruling out

our hypothesis of increased short-term net CH_4 uptake after the fire. Variations in net CH_4 uptake were rather dependent on the weather conditions. The growing season 2018 was the coldest and wettest of the three sampling years, coinciding with the lowest overall net CH_4 uptake. Soil net CH_4 uptake decreased significantly with increasing soil moisture (Figure 6a), emphasizing soil moisture as a controlling factor for atmospheric CH_4 oxidation in upland soils (Whalen & Reeburgh, 1996). In our study, net CH_4 uptake was not correlated to soil temperature, which is in contrast to other studies suggesting soil temperature to be a controlling factor of net CH_4 uptake in upland tundra soils (D'Imperio et al., 2017; St Pierre et al., 2019). Boreal forest fire studies that observed increased CH_4 uptake in boreal forests after fire mostly linked it to changes in soil conditions mainly increased soil temperature and decreased soil moisture (Koster et al., 2017; Kulmala et al., 2014; Morishita et al., 2015; Song et al., 2017; Takakai et al., 2008). However, soil moisture and soil temperature in our study were not affected by the experimental fire (Figure 5), beyond the fire treatment itself, but were influenced by weather conditions. This suggests that in well-drained Arctic soils a typical tundra fire event most likely will not cause major changes in soil CH_4 uptake, but that year-to-year variations are more dependent on weather patterns.

4.5 | Immediate and short-term effects of the experimental fire on N_2O fluxes

The overall very low N_2O fluxes observed in this study (Figure 4e; Table 1), are in line with the range of fluxes reported for upland tundra (Voigt et al., 2020). We hypothesized an immediate increase of N_2O fluxes after the fire due to increases in NH_4^+-N (and NO_3^--N) associated with damage of roots and markedly reduced plant uptake of inorganic N. On the contrary, and despite significantly higher soil concentrations of NH_4^+-N and NO_3^--N immediately after the experimental fire (DAF5), N_2O emissions in burnt plots did not increase but rather decreased when compared to control plots, taking into account the whole growing season immediately after the fire (Figure 4e). This was in line with the few studies on the effects of fire on N_2O fluxes in permafrost regions, reporting either decreased (Kim & Tanaka, 2003; Koster et al., 2017; Takakai et al., 2008) or no effect of fire on N_2O fluxes (Morishita et al., 2015; Ribeiro-Kumara et al., 2020). Reduced N_2O emissions have been linked to lower soil moisture content or too high soil temperatures during fires (Kim & Tanaka, 2003; Koster et al., 2017). However, we did not observe any difference in soil moisture between burnt and control plots. Thus, the lower N_2O emissions in burnt plots immediately after the fire could have been due to a temporary decrease in microbial activity because of heat stress (Barreiro et al., 2020). Additionally, Koster et al. (2017) suggested a possible limiting effect of fire-derived charcoal, on the soil nitrification processes, which is the predominant N_2O production process in aerobic, well-drained tundra soils (Butterbach-Bahl et al., 2013; Firestone & Davidson, 1989; Voigt et al., 2020). Two and 3 weeks after the fire, soils in both control and burnt plots were net N_2O sinks.

There was no short-term effect of fire on N_2O fluxes, supporting the argument of a temporarily fire-affected decrease in microbial activity immediately after the fire. In the first (2018) and second (2019) growing seasons after the fire soils continued fluctuating between being a net source and a net sink of N_2O. This has been observed in other well-drained upland tundra soils, although consumption of N_2O through denitrification seemed unlikely in aerobic conditions (Brummell et al., 2014). Terrestrial net fluxes of N_2O are a fine balance between N_2O production and consumption processes in soils (Butterbach-Bahl et al., 2013). Observed N_2O fluxes did not correlate with soil moisture and only in 2017 did N_2O emissions from burnt and control plots correlated positively with soil temperature. The processes controlling the N-cycle and the involved interactions with biotic and abiotic factors are yet poorly understood (Voigt et al., 2020). The assessment of fire effects on N_2O fluxes is therefore difficult, especially since the indirect effects through soil moisture and soil temperature may be masked by the complex interactions (Ribeiro-Kumara et al., 2020; Voigt et al., 2020).

4.6 | Impacts of warming by OTCs

The significant increase in soil temperatures (0.9–1.3°C; Figure 5b) due to warming by OTCs is within the range of the comprehensive evaluation reports of the OTC's warming effects (Healey et al., 2016; Henry & Molau, 1997). The lack of significant warming effect on the observed soil moisture can partly be related to high spatial variation but may also indicate that the warmer air inside the OTCs did not sufficiently outweigh the loss of energy e.g. through lateral hydrological processes (Hobbie & Chapin, 1998).

Projections from climate models are very robust regarding a general increase in air temperatures within decades, but marked uncertainties are related to future precipitation changes (Meredith et al., 2019). Therefore, although OTCs primarily affected soil temperatures within a small area, the observations give valuable indications of how climate change in form of increased temperatures may influence post-fire carbon and nutrient cycling in an Arctic dry tundra ecosystem.

The overall significant increase in soil temperature by warming through OTCs led to significant short-term increases of ER and GEP in control plots in the growing seasons 2018/19 (Figure 4b,c). This warming effect in an Arctic upland ecosystem has been reported in other studies and linked to increasing temperatures stimulating either microbial activity and carbon decomposition rates or higher autotrophic respiration due to higher primary production (Hicks Pries et al., 2015; Natali et al., 2015; Pedersen et al., 2017; Ravn et al., 2020).

Although soil temperatures were significantly higher in burnt OTC plots compared to burnt no-OTC plots in 2018 (but not in 2019), we did not observe any significant effects of warming in burnt plots CO_2 fluxes. This is in contrast to a study from an Alaskan boreal forest (organic layer <5 cm), observing warming by OTCs significantly increased soil respiration 1 and 2 years after a wildfire,

most likely due to increased microbial activity (Bergner et al., 2004). However, Allison et al. (2010) did not observe any significant warming treatment effects on soil respiration, 6–8 years after the fire in the same study area. This suggests soil temperature is not the only controlling factor for post-fire respiration. Another factor could be that fire may lead to decreased substrate availability and thus counteract the warming-induced microbial activity (Allison et al., 2010; Knicker, 2007). A reduction of decomposable C in our burnt soils reported by Xu et al. (2021), may explain the lack of warming treatment effect on ER in our burnt soils. Additionally, we did not observe any significant differences in GEP between no-OTC and OTC burnt plots, and thus likely no differences in autotrophic respiration.

The lack of significant differences in NEE in burnt OTC compared to burnt no-OTC plots suggests that a typical Arctic dry heath fire under warmer conditions likely does not turn the ecosystem into a bigger CO_2 source compared to a fire in ambient temperatures. However, unburnt plots with OTCs had significantly higher net CO_2 uptake (NEE) due to increased photosynthetic activity, indicating fire turned burnt plots into a significantly bigger net CO_2 source compared to unburnt ecosystems under warmer conditions. On the other hand, Xu et al. (2022) observed a significant increase in top soil (0–3.5 cm) root biomass in burnt OTC plots already 1 year after the fire. This implies a potentially faster recovery of fire-affected ecosystems in a warmer climate, than compared to the observed 4–5 years in another post-fire tundra ecosystem in ambient conditions (Bret-Harte et al., 2013; Jiang et al., 2015). The observed GEP and NEE fluxes in burnt OTC plots 2 years after the experimental fire, however, do not point toward a sustainable and faster recovery of aboveground vegetation in burnt soils under warmer conditions.

Observations of net CH_4 uptake rates 1 and 2 years after the experimental fire from the burnt plots did not show any temperature sensitivity of soil CH_4 oxidation through methanotrophs as observed in the control plots (growing season 2018; Figure 4d) and other Arctic dry heath tundra soils (Lau et al., 2015; St Pierre et al., 2019). However, as previously described soil diffusivity through soil moisture primarily controls soil net CH_4 uptake (Whalen & Reeburgh, 1996). Therefore, the non-significant higher soil moisture levels in burnt OTC plots may have counteracted the warming effect. Another possibility is the fire-induced reduction of soil pore volume by ash or changes in the microbial community (Kennard & Gholz, 2001; Kulmala et al., 2014).

Fluxes of N_2O in burnt soils did not show any temperature sensitivity (Figure 4e), despite significantly higher NO_3-N concentrations in burnt OTC plots. Although higher NO_3-N availability suggests favorable conditions for N_2O production through denitrification processes in water-filled microsites (Dannenmann et al., 2018; Xu et al., 2021), the soil moisture levels could have been too low to create the needed anaerobic conditions in the soil environment. Similar to burnt soils, there was no warming effect on N_2O fluxes in control plots. This is in line with another study from an upland tundra ecosystem, observing contrasting directions and responses of N_2O fluxes to warming by OTCs (Kolstad et al., 2021). This stresses once more the complexity of N_2O processes in Arctic tundra soils.

Overall the experimental fire seemed to have decreased temperature sensitivity for all GHG fluxes compared to the temperature sensitivities in an undisturbed ecosystem.

5 | CONCLUSION

This study addresses the knowledge gap of immediate and short-term effects of experimental fire on GHG exchange in an Arctic upland tundra ecosystem. The combustion of aboveground vegetation by the fire turned burnt areas into net CO_2 sources throughout the entire study period. In burnt areas, the effect of an increase in net CO_2 release was mainly linked to decreased photosynthetic activity and an immediate increase in ER. Even when combined with warming, neither photosynthetic activity nor ecosystem respiration increased in burnt plots, despite the significant temperature increase of 1°C by OTCs. Although the experimental and relatively low-severity fire was a destructive force on the aboveground, it only had a limited impact immediately after the fire on the belowground properties and variations in processes over seasons and between years, which were rather driven by weather conditions. Thus, in a well-drained Arctic tundra ecosystem, the short-term effects of a low-intensity fire event on belowground associated GHG processes, such as CH_4 and N_2O consumption and production are likely to be negligible. Overall, variances of net soil uptake of CH_4 in ambient as well as warmer conditions were primarily controlled by soil moisture levels. Destruction of vegetation by the fire and the subsequent inorganic N flush did not have the hypothesized stimulatory effect on the generally low N_2O emissions, but immediately after the fire N_2O fluxes significantly decreased. However, already 1 and 2 years there was no fire effect on N_2O fluxes neither with nor without warming. The total GHG budget, in CO_2-eq fluxes in this well-drained Arctic tundra ecosystem emphasizes the predominant contribution of CO_2 efflux following the fire. An important factor in the longer-term effects of a typical fire is therefore the rate of plant recovery and their uptake of CO_2. In this study, we addressed the effects of a typical fire in a well-drained tundra with a shallow organic layer. Many wildfires cause both low and highly severely burnt areas, particularly in ecosystems with deeper peat formation, which may lead to greater effects on the net GHG budget. Thus, further research on the effect of high-severity and high-intensity fires on fluxes of all three GHG in an upland Arctic tundra as well as in other Arctic ecosystems is needed.

CONFLICT OF INTEREST
The authors declare no conflict of interests.

DATA AVAILABILITY STATEMENT
The data that support the findings of this study are openly available in Dryad at 10.5061/dryad.v9s4mw6zj.

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