

Wildfire overrides hydrological controls on boreal peatland methane emissions

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Abstract. Boreal peatlands represent a globally important store of carbon, and disturbances such as wildfire can have a negative feedback to the climate. Understanding how carbon exchange and greenhouse gas (GHG) dynamics are impacted after a wildfire is important, especially as boreal peatlands may be vulnerable to changes in wildfire regime under a rapidly 10 changing climate. Yet, given this vulnerability, there is very little in the literature on the impact such fires have on methane (CH_4) emissions. This study investigated the effect of wildfire on CH_4 emissions at a boreal fen near Fort McMurray, AB, Canada, that was partially burned by the Horse River Wildfire in 2016. We measured CH_4 emissions and environmental variables (2017-2018) and CH_4 production potential (2018) in two different microform types (hummocks and hollows) across 15 a peat burn severity gradient (unburned (UB), moderately burned (MB) and severely burned (SB)). Results indicated a switch in the typical understanding of boreal peatland CH_4 emissions. For example, emissions were significantly lower in the MB and SB hollows in both years compared to UB hollows. Interestingly, across the burned sites, hummocks had higher fluxes in 2017 than hollows at the MB and SB sites. We found typically higher emissions at the UB site where the water table was close to the surface. However, at the burned sites, no relationship was found between CH_4 emissions and water table, even under similar hydrological conditions. There was also significantly higher CH_4 production potential from the UB site than the burned sites. 20 The reduction in CH_4 emissions and production in the hollows at burned sites highlights the sensitivity of hollows to fire, removing labile organic material for potential methanogenesis. The previously demonstrated resistance of hummocks to fire also results in limited impact to CH_4 emissions and likely faster recovery to pre-fire rates. Given the potential initial net cooling effect resulting from a reduction in CH_4 emissions, it is important that the radiative effect of all GHG following wildfire across peatlands is taken into account.

25 1 Introduction

Northern peatlands are an important component of the global carbon (C) cycle, acting as long-term sinks of atmospheric carbon dioxide (CO_2). They are also large sources of methane (CH_4) (Bridgman et al., 2013), with northern peatlands contributing approximately 40 – 155 Tg to global CH_4 emissions (Neef et al., 2010; Turetsky et al., 2014). CH_4 dynamics in peatlands results from a combination of various biogeochemical processes (Lai, 2009). Controls on CH_4 production, oxidation and 30 emissions include microtopography (Cresto Aleina et al., 2016), water table depth (Bubier et al., 1995; Granberg et al., 1997), soil temperature (Granberg et al., 1997; Saarnio et al., 1998), substrate quality and availability (Granberg et al., 1997; Segers, 1998; Joabsson et al., 1999), and vegetation cover (Ström et al., 2005; Strack et al., 2017).

However, disturbances such as wildfire can have a significant impact on the magnitude of C fluxes across peatlands (fire can release between 10-85 kg C m⁻² through combustion and smouldering; Turetsky et al., 2011), potentially causing a negative feedback to the climate (Randerson et al., 2006). Western boreal Canada is undergoing increasing pressure from wildfire, with fire extent and frequency expected to double by the end of this century (Benscoter et al., 2005; Flannigan et al., 2008).

5 Understanding how ecosystem C cycling and greenhouse gas (GHG) dynamics are impacted after a wildfire is important, especially as boreal peatlands may be vulnerable to changes in wildfire regime under a rapidly changing climate (Flannigan et al., 2008). Fire can remove surface vegetation, increasing net radiation at the ground surface (Brown et al., 2015), and can ‘reset’ vegetation communities back to the primary succession stage (Johnstone, 2006; Benscoter & Vitt, 2008). Fire can alter soil organic matter quality in the soil column (Neff et al., 2005; Olefeldt et al., 2013a) and reduce belowground C stores in

10 peatlands (Wilkinson et al., 2018). Overall, wildfire can lead to a decrease in C accumulation rate through combustion loss, reduction in vegetation productivity and increased organic matter decomposition post-fire (Ingram et al., 2019; Robinson & Moore, 2000; Wieder et al., 2009). Furthermore, increased ash deposition after wildfire can increase soil pH (Molina et al., 2007; Davies et al., 2013) and change the physical characteristics of the soil, including blocking of peat macropores and altering hydrology (Noble et al., 2017). Reduction in vegetation cover and soil organic matter can also lead to drier conditions across

15 peatlands (Tarnocai, 2009; Thompson & Waddington, 2013; Kettridge et al., 2015), with the drop in water table level causing an increase in the aerobic zone (Waddington et al., 2015). This could lead to a reduction in CH₄ emissions or even uptake of CH₄ via oxidation (Strack et al., 2004; Turetsky et al., 2008; Moore et al., 2011). Conversely, high water tables can occur post-fire (Kettridge et al., 2015), although often associated with low surface moisture contents due to hydrophobicity of the peat (Doerr et al., 2000). Low soil moisture rates can also occur under increased ash deposition after fire, with increased closure of

20 soil pores by ash causing reduced capacity to hold water and increased runoff (Heydari et al., 2017).

Microtopography (microforms) across peatlands can be impacted through fire, by increasing the prominence of hollows (low lying areas close to the water table; Belyea & Clymo, 1983) on the landscape through altering elevation (Benscoter et al., 2015), and often hollows will have a higher severity of burn compared to other areas across the landscapes (Mayner et al., 2018; Benscoter et al., 2005). Conversely, hummocks (mounded microtopography, approximately 0.2 m or higher above the

25 water table; Belyea & Clymo, 1983) are generally resistant to fire, namely due to moisture retention differences between the different moss species present at both microform types, as *Sphagnum* spp. is much more resilient to fire than feather moss (Kettridge et al., 2015).

Despite the increasing pressures from wildfires across northern peatlands, a knowledge gap still persists on CH₄ emissions after wildfire, especially in boreal regions. In a study on the impact of wildfire on methanotrophic communities from an

30 ombrotrophic peat bog, Danilova et al. (2015) found a reduction in the activity of the methanotrophs in burned sites 7 years post-fire. This reduction following wildfire could therefore lead to a potential increase in CH₄ emissions from bog systems. Grau-andrés et al. (2019) also showed an increase in CH₄ emissions at an ombrotrophic bog in the UK one year after a prescribed fire, most likely due to increased graminoid coverage. Conversely, studies at other bog sites in the UK report a decrease in emissions after fire (Ward et al., 2007; Davies et al., 2013). In non-peatland ecosystems across boreal regions,

wildfire has been shown to cause an increase in CH₄ uptake (Burke et al., 1997; Song et al., 2017, 2018). However, in permafrost zones, wildfires can often typically lead to substantial permafrost thaw and increasing moisture levels across the landscape (Gibson et al., 2018), potentially leading to an increase in CH₄ emissions (Kim & Tanaka, 2003; Turetsky et al., 2008; Olefeldt et al., 2013b; Helbig et al., 2017). However, Köster et al. (2017; 2018) found an increase in CH₄ uptake across 5 continuous permafrost sites in both Canada and Russia after fire. To date, we cannot find any reference on the impact of fire on CH₄ emissions across fens, despite being the dominant peatland type in western boreal Canada (Vitt et al., 2000). Therefore, the objectives of this study are to: i) determine the impact of wildfire on fen CH₄ emissions across a peat burn 10 severity gradient; ii) evaluate the controls on CH₄ emissions within each site; and iii) examine CH₄ production potential across a peat burn severity gradient. We hypothesize that CH₄ emissions and production will be lower at burned sites due to lowering of the water table, changes in substrate availability, and reduction in vegetation cover.

2 Methods

2.1 Study site and collar locations

The study was undertaken in a treed moderate-rich fen (hereafter referred to as Poplar Fen), 20 km north of Fort McMurray, Alberta, Canada (56°56.330 N, 111°32.934 W), which was partially burned by the Horse River Wildfire in 2016. The mean 15 annual temperature (1981-2010) is 1 °C, and mean annual precipitation is approximately 420 mm (Environment Canada, 2017). This treed fen is dominated by *Larix laricina* (Du Roi) K.Koch, *Picea mariana* (Mill.) Britton, *Betula pumila* (L.), *Equisetum fluviatile* (L.), *Smilacina trifolia* (L.) Sloboda, *Carex* spp. and *Sphagnum fuscum* (Schimp.) Klinggr and brown mosses, largely *Tomenthypnum nitens* (Hedwig) Loeske. All vegetation was identified to the species level, with nomenclature for vascular plants and mosses as Flora of North America Editorial Committee (1993). Average peat depth ranges between 1 and 1.5 m. 20 The landscape consists of approximately 47% hummocks and 53% hollows within the fen area (Gabrielli, 2016). This site was split into three sections along a peat burn severity gradient as assessed by depth of burn (DOB). The unburned (UB) site was situated within the fen interior and was not affected by the wildfire. The moderately burned (MB) site had a DOB of approximately 9 – 11 cm and the severely burned (SB) site had a DOB of approximately 14.5 – 17 cm. Both burned sites were situated on the peatland margin, closer to the adjacent upland at a slightly higher elevation. The DOB was determined following 25 the protocols used by Lukenbach et al. (2015a). In summary, this method assumes a pre-fire flat surface between multiple reference points across the site, including adventitious roots in the burned sites and unburned reference points. A string is attached between two reference points and ten measurements were taken along the length, from string to burned ground surface, giving an estimate of the depth of the burn. At each site, PVC collars (height 15 cm x diameter 20 cm) were placed in both hollows and hummocks. A total of four replicate collars were placed at each microform location at each site to a depth of 30 approximately 15 cm in Spring 2017, totaling eight collars at the UB site, eight collars at the MB site and eight collars at the SB site. The height of the collar was measured from the soil surface in order to have the correct chamber headspace volume for CH₄ emission calculations.

2.2 Environmental Conditions

Water table (WT) depth (relative to the ground surface) was measured adjacent to each pair of collars at all three sites. A PVC pipe (4 cm (diameter) x 100 cm (length)) fully slotted along the full length and covered in mesh was used. A soil temperature (ST) profile was collected at each collar during each CH₄ measurement at -30, -25, -20, -15, -10, -5, -2 cm from ground surface.

5 Percentage cover of plant functional type (bryophyte and over story graminoid and dwarf shrub), as well as bare ground, burned material and standing water was estimated from photographs taken at peak growing season during 2018 to produce percentage cover estimates of the flux collars.

2.3 Measurements of field CH₄ emissions

10 Methane fluxes were measured using the closed chamber method, eight times between 7th May and 16th August 2017 and 14 times between 11th May and 16th August 2018. A cylindrical opaque chamber (20 cm x 50 cm) was placed on the collar, with water poured around the collar edge to create a seal. A battery powered fan was used to mix the chamber headspace. A thermocouple located within the chamber, attached to a thermometer was used to measure temperature during sampling. A 20 mL syringe was used to collect gas samples at intervals of 7, 15, 25 and 35 minutes following chamber closure and injected

15 into Exetainers (Labco, UK). A gas chromatograph (GC; Shimadzu GC2014, Mandel Scientific, Canada) with a flame ionization detector (250 °C), helium gas carrier and standards of 5 and 50 ppm was used to determine CH₄ concentration of the gas samples collected during the field seasons. The emissions were determined from the linear change in concentration over time, which includes corrections for temperature and volume of the chamber. Any small negative or positive values in which the change in concentration did not exceed 10% (precision of concentration analysis) were assigned a zero-emission

20 value. Large negative emissions (< -5 mg CH₄ m² d⁻¹) were removed from the analysis as it is unlikely this system would have consumption rates as large as this and were likely caused by disturbance during chamber placement. These procedures resulted in a total loss of 6 % of the data across both years. In order to determine whether emissions measured from the UB site were representative of emissions from Poplar Fen as a whole, we also compared our fluxes to a previous study of CH₄ emissions collected between 2011 and 2014 at the fen.

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2.4 Potential CH₄ production

Peat samples were collected adjacent to each collar (2-3 m away to avoid disturbing the collar) on the 13th August 2018 and immediately shipped to the laboratory and frozen until analyzed. Peat samples were obtained using a tin can to a depth of 20 cm from the ground surface (two samples were collected per sampling location; 0 – 10 cm and 10 – 20 cm).

30 Potential CH₄ production was determined under anaerobic conditions following a similar methodology to Strack et al. (2004). Peat slurries of approximately 20 g of wet peat were made in 250 mL incubation jars. Distilled water was added to the sample to saturate, without allowing for standing water. Samples were flushed with N₂ for 15 minutes and then sealed. Slurries were incubated at room temperature (approximately 20°C) and sampled at 0, 24, 48 hr and then twice weekly between 16th October

2018 and 2nd November 2018 (total incubation length of 17 days). Samples were agitated by hand before sampling commenced to mix the gases within the peat pore spaces and the jar headspace.

Samples (10 mL) were extracted from the jars and injected into a Fast Methane Analyzer (FMA; Los Gatos, USA). A 10 mL sample of N₂ was replaced in each jar after the sample was extracted to maintain headspace pressure. Potential methane production was determined from the linear increase in CH₄ concentration within the jars over the incubation period after 5 correcting for dilution by N₂ (Strack et al., 2017). Gravimetric soil moisture (GWC) was determined by weighing subsamples of peat not used in the incubation, drying the samples at 60°C for 2-3 days and reweighing. Organic matter content was determined by Loss on Ignition (LOI), burning samples at 550°C for 4 hours.

10 **2.5 Data analysis**

All statistical analysis was undertaken in R (R Core Team 2013) using the package *nlme* (Pinheiro et al., 2018), and all output and models were inspected for normality and homogeneity of residuals (Zuur et al., 2009). Data were log transformed if required, with a value of ten being added prior to transformation to account for zeros and negative values in the dataset. Seasonal mean values of CH₄ flux and associated environmental variables at each collar were used in all model levels (Treat 15 et al., 2007; Turetsky et al., 2014) with statistical significance considered at the $\alpha = 0.05$.

In order to evaluate the effect of burn severity on CH₄ flux, a linear mixed effects model (LMM) was used with burn severity, microform, the two-way interactions between these, and year as fixed factors, with collar ID as a random effect to take into account repeated measures (Pinheiro et al. 2018). Another LMM was used to evaluate the environmental controls on CH₄ flux, with the effect of burn severity, WT, ST at 30 cm depth and the two-way interactions between each included as fixed effects.

20 If significant factors were found, Tukey pairwise comparisons were completed using the *lsmeans* package (Lenth, 2016). Any insignificant factors were removed from the model until the final model was found. An individual insignificant factor was kept in the model if its interaction with another factor was significant. The amount of variance explained by the model (R^2_{GLMM}) was calculated using the method described by Nakagawa & Schielzith (2013).

A two-way Analysis of Variance (ANOVA) was used to test for differences between CH₄ production rate, burn severity and 25 microform. No significant difference between sampling depth was found, thus depths were combined in further analyses. Again, if significant differences were found, Tukey pairwise comparisons were completed.

3 Results

3.1 Environmental and vegetation variables

Water table depth was linked to microtopographic position, with hollows having highest WT position across all sites. The 30 deepest WT depth (\pm standard deviation) during both 2017 and 2018 was found at the SB hummocks, approximately -45 ± 5.6 and -44 ± 6.0 cm below the surface respectively (Table 1). The shallowest WT was found at the unburned hollows, being -8.6 ± 4.1 and -6.9 ± 1.8 cm below the surface in 2017 and 2018, respectively (Table 1). Average ST at 30 cm depth (taken as spot

measurement during each gas flux measurement) was similar across the burn severity gradient at approximately 10 – 12 °C (Table 1).

The vegetation survey of the collars undertaken in 2018 indicated that bryophytes dominated across all collars at all sites, regardless of microtopographic position. UB hollows were dominated by the moss *T. nitens*, while the hummocks were 5 dominated by *S. fuscum*. The SB hummocks and hollows both had the highest percentage of bare ground at ~ 49% and ~ 55% cover respectively, indicating vegetation (mostly *T. nitens*) was completely removed during the fire. It was noted that *Polytrichum strictum* Bridel, J. Bot (Schrader) moss was beginning to colonise these bare areas. The MB hummocks had the highest percentage of burned material (~ 40 % cover; predominantly singed *S. fuscum*) (Figure 1).

10 3.2 CH₄ emissions

The average CH₄ flux (\pm standard deviation) at unburned hollows was 126.5 ± 80.5 and 56.3 ± 18.9 mg CH₄ m⁻² d⁻¹ in 2017 and 2018, respectively (Figure 2). CH₄ emissions were much lower in the MB and SB hollows in both years, with the average flux being -0.38 ± 1.6 and -0.46 ± 0.9 mg CH₄ m⁻² d⁻¹ in 2017 and 0.21 ± 1.7 and 0.62 ± 2.5 mg CH₄ m⁻² d⁻¹ in 2018 (Figure 2), respectively. Interestingly, across the burned sites, hummocks had higher fluxes in 2017 than hollows with the average flux 15 being 1.10 ± 2.04 mg CH₄ m⁻² d⁻¹ at the MB site and 4.53 ± 9.3 mg CH₄ m⁻² d⁻¹ at the SB site (Figure 2). During 2018, fluxes were lower, with hummocks being a slight sink of CH₄ with average fluxes of -0.18 ± 2.06 mg CH₄ m⁻² d⁻¹ at the MB site and 0.43 ± 1.6 mg CH₄ m⁻² d⁻¹ at the SB site (Figure 2). No significant difference in emissions was found [$t(22.5) = 1.5$, $p = 0.154$] between the present study and the study undertaken between 2011 and 2014 (Figure S1).

Results of the LMM illustrate that there was a significant effect of burn on CH₄ flux (Table 2), but no significant effect of 20 microform or year (Table 2). The second LMM considering environmental controls explained 63% of the variance in CH₄ emissions and found a significant interaction between burn severity and WT depth (Table 2; Figure 3), but no significant relationship between CH₄ flux with WT depth, ST (Figure S2) or burn severity alone.

3.3 Potential CH₄ production

25 Measured potential CH₄ production was highest in the unburned hollows ranging from between 0.006 and 0.13 µg g⁻¹ peat hr⁻¹ (Figure 4); however, there was no significant effect of burn severity [ANOVA, $F = 2.959$, $p = 0.065$]. CH₄ production was much lower across the burned sites, ranging between 0.0001 and 0.004 µg g⁻¹ peat hr⁻¹. The MB hummocks followed a similar pattern to the field measurements of CH₄ flux, having higher potential CH₄ production than the hollows. No significant difference in organic matter content or gravimetric water content was found between sites or microform types (Table S1).

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4 Discussion

Fire had a strong effect on CH₄ emissions in this study, causing a large decrease in CH₄ flux in the MB and SB hollows in comparison to the UB hollows. Conversely, this study also highlights the resistance of hummocks to fire (Wieder et al., 2009; Benscoter et al., 2015), with hummocks across the burned sites maintaining higher CH₄ emissions after the fire compared to

hollows. Methane production in the laboratory followed a similar trend to the field study, with highest production in the UB hollows and virtually no production in the burned hollows, again highlighting this reversal of typical peatland CH₄ emissions. These results contrast with other studies looking at CH₄ emissions post-fire at peatland sites, with Danilova et al. (2015) indicating that fire across an ombrotrophic bog could decrease CH₄ oxidation due to removal of the methanotrophic 5 community, while Grau-andrés et al. (2019) note a potential increase CH₄ emissions due to increased graminoid cover. We did not specifically measure CH₄ oxidation in this study.

We hypothesized that the lower CH₄ emissions at the burned sites could be due to the intensity of the burn, reducing substrate availability (labile carbon) and minimizing the methanogenesis community, resulting in lower emissions. An increase in fire frequency has the potential to reduce organic matter quality and change vegetation communities in peatlands (Lukenbach et 10 al., 2015b). Associated with a change in vegetation communities is the potential change in biogeochemical cycling and microbial processes (Ward et al., 2007). For example, the slight recovery in CH₄ emissions in 2018 (two years postfire) could be due to vegetation recovery (Ward et al., 2007), providing more available substrate through root exudates for CH₄ production (Greenup et al., 2000; Robroek et al., 2015). The presence of graminoids in the SB hollows post-fire was also found, which could also lead to increasing CH₄ emissions in the future, as plant-mediated transport of CH₄ is well documented across 15 peatland ecosystems (Bellisario et al., 2016).

Higher emissions at the UB site could result from overall shallower WT at this location compared to the MB and SB sites (Table 1), which were located at the fen margins. Poplar Fen has a highly variable connection to groundwater (Elmes et al., 2018) and the hydrogeologic setting of Poplar Fen likely contributed to the limited effect of the wildfire at this location, but could also result in higher CH₄ emissions than would have occurred naturally at the burned sites prior to the fire. However, the 20 comparison of our results to emissions measured between 2011 and 2014 at another location in Poplar Fen burned during the fire indicate there was no significant difference in CH₄ emissions. Interestingly, we see no relationship with CH₄ emissions and WT depth at the burned sites. This switch in the typical understanding of the relationship between CH₄ emissions and WT further strengthens our argument on the overriding influence of fire. Even under suitable hydrological conditions, there is a 25 lack of CH₄ production, as shown in the incubation study. Removal of vegetation and soil organic matter can lead to drier conditions (Thompson & Waddington, 2013), with a lower water table creating a larger aerobic zone, potentially leading to lower rates of CH₄ production and potentially greater rates of CH₄ consumption (Strack et al., 2004; Moore et al., 2011). However, fire can also cause a higher water table, which could potentially lead to larger anaerobic zones and potentially higher 30 CH₄ emissions. However, this is dependent on the severity of the burn, where a low severity fire which only removes vegetation and does not impact the microbial community and organic matter content of the soil may still allow for CH₄ production. Conversely, a high severity burn which potentially has removed these communities and organic matter may no longer allow for CH₄ production, even with suitable hydrological conditions.

The higher CH₄ production found at the MB hummocks is likely due to the small methanogen community surviving the fire, due to the resistance of *S. fuscum* to fire (Benscoter et al., 2011). After fire, there could be chemical changes in the soil substrate, such as an increase in availability of terminal electron acceptors, that could contribute to the reduction in CH₄ production and

emissions (Wilson et al., 2017). Therefore, there is a potential long-term impact on the biogeochemical processes of peatlands (Danilova et al., 2015) and in order to fully understand the overall impact of wildfire on CH₄ emissions, additional studies at other sites encompassing the full range of boreal peatland types would be key. This is especially true given the conflicting results in the literature regarding the overall impact of fire across a variety of peatland sites. Continuous monitoring of the 5 recovery of the ecosystem over time could help evaluate the amount of time required for CH₄ emissions return to similar levels as the undisturbed site.

5 Conclusion

This study investigated the impact of wildfire on CH₄ emissions at a treed, moderate-rich fen in northern Alberta. We believe 10 this is the first study to investigate the impact of wildfire on CH₄ emissions at a non-permafrost boreal fen. The results showed a significant impact of fire on the magnitude of CH₄ flux, with a significant reduction in flux observed at the burned sites in comparison to the unburned (UB) site. No relationship was found with water table at the burned sites, contrasting the significant relationship at the UB site, further illustrating that methanogenesis was limited following fire. This was further supported by 15 a lower rate of CH₄ production from peat collected at burned sites compared to UB, likely linked to reduced methanogen population and/or substrate availability due to the resistance of *Sphagnum* spp. hummocks reducing burn severity. With the expected increase in wildfire frequency across western boreal Canada, it is vital we fully understand the impact of fire on CH₄ dynamics. If fire is to reduce CH₄ emissions and production across these peatland ecosystems, there is a potential initial net 20 cooling effect, therefore it is important to take into account the radiative effect of all GHG following wildfire.

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Code/Data availability

The data that support the findings of this study are available from the corresponding author upon reasonable request

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Author contribution

MS and RP secured the funding; SJD, CVB, RP and MS designed the study; SJD performed the research; SJD analyzed the data with input from MS.; and SJD, CVB, RP and MS wrote the paper.

Competing interests

The authors declare they have no conflict of interest.

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Table 1. Environmental characteristics (mean (\pm standard deviation)) for each microform type across burn severity gradient.

Year	Burn severity ^a	Microform	Water table depth (cm)	Soil temperature at 30 cm depth ($^{\circ}$ C)
2017	UB	Hollow	-8.6 (4.1)	11.0 (1.1)
		Hummock	-19.9 (5.5)	11.5 (3.6)
	MB	Hollow	-20.9 (7.6)	11.2 (3.4)
		Hummock	-37.5 (9.2)	10.6 (1.7)
2018	SB	Hollow	-24.0 (15.3)	10.2 (2.0)
		Hummock	-45.6 (5.6)	9.9 (2.7)
	UB	Hollow	-6.9 (1.8)	12.1 (1.9)
		Hummock	-22.6 (6.6)	12.3 (1.3)
	MB	Hollow	-9.0 (0.9)	11.1 (1.1)
		Hummock	-23.6 (0.2)	12.8 (0.8)
	SB	Hollow	-14.2 (1.0)	10.3 (0.3)
		Hummock	-44.5 (6.0)	11.4 (1.4)

^a UB is unburned, MB is moderately burned and SB is severely burned.

Table 2. Statistical results of the linear mixed effects model ^a

Flux component	Effect	F	P	R^2_{GLMM} ^b
	Burn severity	$F_{1,22} = 83.02$	< 0.0001	
	Microform	$F_{2,17} = 2.4$	0.14	
	Year	$F_{1,22} = 1.2$	0.3	0.52
	Burn severity \times Microform	$F_{2,17} = 5.5$	0.014	
	Intercept	$F_{1,22} = 2075.3$	< 0.0001	
CH_4^c	Burn severity	$F_{2,20} = 83.4$	< 0.0001	
	WT depth	$F_{1,17} = 0.5$	0.5	
	ST at 30 cm depth	$F_{1,17} = 3.0$	0.1	
	Burn severity \times WT depth	$F_{2,17} = 3.3$	0.046	0.63
	Burn severity \times ST at 30 cm depth	$F_{2,17} = 3.4$	0.06	
	Intercept	$F_{1,20} = 2085.4$	< 0.0001	

^a All models have a random factor of collar ID to take into account the repeated measures across both years

^b We report the marginal R^2_{GLMM} accounting for variance explained by fixed factors only

^c The model was calculated using $\log_{10} \text{CH}_4$ values

Figure 1. Vegetation cover (%) for each dominant plant functional type and ground cover for the flux collars in 2018 across the peat burn severity gradient.

5 **Figure 2. Methane (CH_4) emissions at each microform type across the peat burn severity gradient for 2017 and 2018. UB is unburned, MB is moderately burned and SB is severely burned. Note that CH_4 values were log transformed + 10 therefore a value of 1 represents the measured value zero.**

10 **Figure 3. Seasonal mean methane (CH_4) flux at each collar across the peat burn severity gradient plotted against seasonal mean water table (WT) depth. UB is unburned, MB is moderately burned and SB is severely burned. Note that CH_4 values were log transformed + 10 therefore a value of 1 represents the measured value zero.**

Figure 4. Potential methane (CH_4) production across the peat burn severity gradient and microform type. Each microform represents four sample replicates.