Methane Cycling in Northern Peatlands Following Wildfire

by

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Author's Declaration

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

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Abstract

Peatlands are an important component of the global carbon (C) cycle, they operate as long-term global sinks of atmospheric carbon dioxide (CO_2) and sources of methane (CH_4). However, they are becoming increasingly vulnerable to disturbances such as wildfire. Understanding the impact of wildfire on greenhouse gas dynamics is important as the frequency and severity of these fires continues to increase. Loss of labile substrate and methanogenic community is often attributed as the driver behind CO_2 and CH_4 emission reductions from peatland soils post-wildfire. Soil incubations were conducted using samples from both burned and unburned peatlands immediately (Alberta) and 2-years (Ontario) post-fire to measure and compare CH_4 production potential and oxidation. In-situ CH_4 and CO_2 flux measurements were conducted at the Alberta site immediately after fire. Environmental variables such as water table depth, soil temperature and moisture were collected at each site. Soil samples from the Ontario site were also analyzed for phenolic compounds, pH, and electric conductivity.

In both the recently burned and 2-year post fire incubations, lower CH_4 prodution was observed at the burned sites. In-situ field fluxes determined that both ecosystem respiration (ER) and net ecosystem exchange (NEE) was lower and CH_4 flux indicated net CH_4 uptake at the burned site compared to the natural site, immediately post-fire. Overall, this study enhances our understanding of the impacts of wildfire on greenhouse gas dynamics and carbon storage in peatland ecosystems both immediately and 2-years post-burn. This understanding is important for the establishment of peatland carbon budgets in response to

climate change, contributing to the development of accurate and reliable global carbon budgets and climate modelling that can account for the increasing vulnerability of boreal peatlands to fire.

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Chapter 1

Introduction

Peatlands cover a large portion of land in Canada and also store a large portion of carbon and exchange greenhouse gases (GHGs), making them a valuable ecosystem when looking at offsetting the impacts of climate change and preventing further carbon loss. As wildfires continue to increase in both frequency and magnitude in Canada, more peatlands are being impacted. However, there is little research on how wildfire affects peatland carbon storage and GHG cycling. This study aims to investigate the impacts of wildfire on peatland methane (CH_4) cycling within the first few years post-wildfire.

1.1 Peatlands

Peatlands are an important component of the global carbon (C) cycle, they operate as long-term sinks of atmospheric carbon dioxide (CO_2) and sources of methane (CH_4) (Charman, 2002). They cover over 4 million km^2 globally (Holden 2005), and are a dominant feature in the Canadian landscape, covering 13% of land area in Canada, accounting for ¼ of the world's peatlands (Tarnocai, Kettles & Lacelle, 2011). Peatlands perform many other valuable ecosystem services in addition to carbon storage such as water retention, storage and purification, and habitat provision to numerous plant and animal species, many of which are identified as endangered or threatened (Fillicetti et al., 2019). Peatlands also possess social, economic, and cultural value as most of these ecosystems are situated within the traditional territory of Indigenous Peoples (Speller & Forbes, 2021). First Nations, Inuit and Metis people have been conserving, protecting, and living in harmony with peatlands in Canada for

centuries, and recognizing this ecosystem's essential role, they are often referred to as "the breathing lands" (Shulz et al., 2019).

Boreal peatlands form over long periods of time due to climate and/or position of the landscape. Multiple factors lead to the complex processes and feedbacks that result in accumulation of organic matter over thousands of years, which is facilitated by slow litter decomposition. These factors include low mean annual temperatures, poor drainage, and short growing seasons (Clymo, 1984). Variations in decomposability of *Sphagnum* moss species, water movement, productivity of the whole plant community and nutrient concentrations all play a role in the formation of microtopography in peatlands (Eppinga et al., 2006). This microtopography is characterized by the formation of hummocks and hollows, where hummocks are elevated mounds of moss species that were less readily decomposed, and hollows are depressions of more easily decomposed *Sphagnum* species (Belyea, 1996).

Different peatland types exist along a spectrum of environmental conditions and their development is influenced by interactions between slight variations in conditions like vegetation, nutrient concentrations, water table position and fluctuations, groundwater connectivity, and pH (Waddington et al., 2009). In Canada, the dominant peatland types are bogs and fens. A major difference between these peatland types is groundwater connectivity; bogs are disconnected from groundwater and typically receive all water and nutrients from rain, making them ombrotrophic and acidic (Clymo, 1984). In contrast, fens are connected to the groundwater or surface water, which is where they receive mineral and nutrient inputs, making them minerotrophic with a near neutral pH (Clymo, 1984) Other key differences are

that bogs are dominated by *Sphagnum* spp., and their water tables are typically below the surface while fens are dominated by brown mosses and have water tables near or above the surface (Belyea, 1996).

1.2 Peatlands and Wildfire

In Canada, many boreal peatlands are severely impacted by anthropogenic disturbances such as oil and gas exploration and extraction, and horticultural peat extraction (Rochefort et al., 2022). These disturbances lead to long-term ecosystem degradation, destruction, and loss of ecosystem functions. Further, under a rapidly changing climate, projected to become warmer with more extreme weather events due to increasing temperatures and unpredictable rainfall patterns, natural disturbances are predicted to increase in frequency and magnitude (Kasischke & Turetsky, 2006). Therefore, boreal peatlands are facing an increased risk of further or initial disturbance. A major concern from a global climate perspective is the impacts of these disturbances on the production and emission of greenhouse gases (GHG), specifically carbon dioxide (CO_2) and methane (CH_4), which would further contribute to climate warming.

One type of peatland disturbance that is likely to become more frequent under a changing climate is wildfire. A disturbance like wildfire has an immense impact on carbon storage and emissions in an ecosystem. While peatlands are resilient to many types of natural disturbances, the effects of increased severity and frequency of wildfires on peatlands is unknown. Through the initial combustion of peat and then smouldering, it was found that between 10 to 85 kg C m^{-2} can be released during a wildfire (Lukenbach et al., 2015).

1.3 Peatland Methane (CH_4) Dynamics

As a result of waterlogged and therefore anoxic conditions, peatlands are typically a source of CH_4. Methane is a potent greenhouse gas with a global warming potential greater than CO_2 by 80.8 times over a 20-year time horizon and 27.3 times over a 100-year time horizon (IPCC, 2021). Methanogens are a group of *Archaea* that produce CH_4, while methanotrophs are *Bacteria* that take up CH_4 through oxidation. Methanogens can only survive in environments with very little to no oxygen, therefore they live within the anoxic zones of peatlands (Garcia et al., 2000). They are most active near the water table where the boundary between the oxic and anoxic zone exists and where oxygen is limited while organic matter substrate quality remains relatively high (Lai, 2009). The CH_4-cycling microbial community in peatlands typically has a significantly greater potential CH_4 oxidation than the potential CH_4 production rate, when the ecosystem is functioning as normal (Dedysh, 2002). This process consists of the methanotrophs oxidizing a large portion of the CH_4 produced in the oxygen-depleted zones, limiting the amount of CH_4 that is released into the atmosphere. Wildfire likely alters microbial community composition and therefore CH_4 dynamics, Danilova et al. (2015) investigated the impacts of wildfire disturbance on the microbial community in a peat bog post-wildfire. They determined that the long-term impacts of wildfire was severe as the structure and function of microbial communities changed substantially. Overall, microbial degradation of organic matter in the burned sites measured in the study rapidly increased 7-years post-fire, resulting in much higher CH_4 emissions than in an undisturbed peat bog due to a lack of oxidation taking place because of the destruction of the methanotrophic community during the wildfire (Danilova et al., 2015). Conversely,

another study that investigated changes in microbial communities post-wildfire revealed that CH_4 oxidation post-wildfire actually increased and that the composition of the methanogenic community did not change (Jaatinen, 2004). A soil bacterial and fungal community study investigated the potential impacts of fire, vegetation, moisture, pH and C when predicting community composition. This study found that fire occurrence was a significant predictor of soil microbial community composition because burned communities are very dissimilar to unburned communities and this dissimilarity increases with increased burn severity (Whitman et al., 2019). It was also found using globally abundant taxa that there were bacterial taxa that were identified as significant fire responders as they were between 35-64x more abundant in burned communities, including *Massilia* sp. and *Arthrobactor* sp. (Whitman et al., 2019). Because of the limited number of studies that analyse the changes in the composition and function of microbial communities caused by wildfires, it is difficult to make firm conclusions regarding the long-term impacts of fire on microbial communities in peatlands.

Microbial function post-wildfire may also be altered when investigating the potential effects of microbial community changes on peat decomposition. CH_4 production potential and methanogen abundance is often highest below the water table where they have access to carbon sources from litter decay and anoxic conditions (Marti et al., 2015). Typically, CH_4 oxidation is more present just above the oxic-anoxic boundary where oxygen (O_2) and CH_4 are more available (Clymo and Bryant, 2008). However, fire can result in the alteration of carbon substrate availability, which is an important factor controlling CH_4 production potential and oxidation as well as methanogen and methanotroph abundance (Sun et al.,

2012; Reumer et al., 2018). Wildfire also alters composition of soil carbon through the creation of charcoal and ash. An incubation study using peat samples from a black spruce peatland found that the addition of ash post-fire may have resulted in the promotion CH_4 production (Hogg et al., 1992).

1.4 Changes in Environmental Conditions Post-Wildfire and their Impact on CH_4 Flux

It has been well-established in the literature that various environmental variables such as temperature and moisture content have significant impacts on CH_4 dynamics in peatlands (Lai, 2009;Turetsky et al., 2014). Water table position is an important environmental variable that impacts CH_4 oxidation and production by changing the size of the oxic and anoxic zones (Lai, 2009). Changes in the water table position due to wildfire disturbance can result in either an increase or decrease in CH_4 emissions, depending on how the hydrology of the peatland was altered (Davidson et al., 2019). Water tables can rise post-fire due to loss of surface peat to combustion, but post-fire conditions are typically also associated with low surface moisture contents (Kettridge et al., 2015), potentially resulting in lower CH_4 emissions. Conversely, water tables can also be deeper post-wildfire, dependent on the severity of the fire and subsequent weather conditions, resulting in lower emissions (Davidson et al., 2019). Due to the variable nature of water tables in peatlands post-wildfire, the long-term impacts on CH_4 emissions post-wildfire may greatly vary depending on the original, pre-disturbance hydrology of the site and the burn severity.

Changes in other environmental variables such as soil temperature, soil moisture, and substrate and nutrient availability also have impacts on CH_4 emissions (Xu et al., 2020). These environmental variables are often severely altered during and post-wildfire (Lukenbach et al., 2015). A reduction in substrate availability as a result of burning and reduction of the methanogenic community will result in a reduction of CH_4 emissions post-wildfire (Davidson et al., 2019). Increases in soil temperature during the fire and post-fire, may increase CH_4 emissions if the methanogenic community is still present, and was not destroyed during the initial burning (Jaatinen, 2004). Increased soil temperatures in peatlands post-wildfire are most likely a result of increased net radiation due to the loss of vegetation during fire (Lukenbach et al., 2015). There are limited conclusions to be made on how CH_4 emissions will be impacted as a result of changes in environmental conditions in peatlands post-wildfire as the literature reviewed presents conflicting findings.

1.5 Impacts of Wildfire-Generated Charcoal on CH_4 Emissions

Although studies on the effects of charcoal on peatland carbon cycling are limited, so insights can be gained from biochar soil amendment. Biochar is a carbon rich by-product of the burning of biomass. The addition of biochar to soil can incur multiple benefits that include increasing carbon storage, decreasing greenhouse gas emissions, improving soil water holding capacity, changes in soil texture to increase aeration, and the addition of phenolics and nutrients (Yu et al., 2012). In multiple studies, it was found that the application of biochar generally resulted in the reduction of CH_4 emissions, as well as significantly

increased pH (Yu et al., 2013; Ji et al., 2018), suggesting that charcoal produced during wildfire, may have a similar effect.

Most studies attribute the reduction in CH_4 emissions in peatlands post-wildfire to the loss of methanogenic community and labile substrate rather than the addition of charcoal (Lai, 2009; Jaatinen 2004; Gray et al., 2020). However, the impacts of charcoal on CH_4 emissions in peatlands post wildfire is not well understood. A recent study investigated the role of pyrogenic carbon introduced to peatlands post-wildfire and determined the important role pyrogenic carbon plays in the reduction of CH_4 production (Sun et al., 2021). The electron snorkel mechanism in pyrogenic carbon facilitates extracellular electron transfer and stimulates alternative microbial respiration that suppress methanogenic activity (Sun et al., 2021).

In a New Zealand study, the amendment of volcanic pumice soil to natural gas leaks, livestock housing, and coal mine vents was used to mitigate CH_4 emissions by supporting CH_4 oxidation (Sayed, 2016). These findings support a hypothesis that charcoal generated by wildfire in peatlands should support CH_4 oxidation as well, increasing the ecosystem's potential as a carbon sink, or at least off-setting some of the emissions from the initial burning of the ecosystem. Another study on freshwater islands in Northern Sweden established that while net soil CH_4 oxidation post-wildfire increased, it was mainly attributed to the changes in the vegetation community, rather than the presence of charcoal (McNamara, 2015).

The addition of charcoal is also suspected to increase phenolic compound concentration through leaching, which could reduce decomposition through the enzymic latch mechanism (Yu et al., 2021; Freeman et al., 2004). Phenol oxidases are responsible for the degradation of phenolic compounds, but low oxygen availability in peatlands allows the accumulation of phenolics that have been shown to reduce the activity of hydrolase enzymes (Yu et al., 2021). Hydrolase enzymes are responsible for organic matter degradation; increases on phenol oxidase activity can release hydrolases from inhibition, further increasing organic matter breakdown, which is known as the "enzymic latch theory" (Freeman et al., 2004). Accumulation of phenolic compounds leached from charcoal, may slow CH_4 production by reducing overall organic matter decomposition through this mechanism. In cases where wildfire induces deeper water table position, this short-term water table lowering can increase decomposition as the presence of O_2 initiates phenol oxidases (Preston et al., 2012). The role that phenolics play in post-wildfire peatland carbon cycling requires further study.

1.6 Objectives

While existing literature has investigated carbon cycling post-wildfire in peatlands, understanding how greenhouse gas dynamics, particularly CH_4 emissions, are impacted in peatlands in the long-term and immediately after wildfire is not well understood. The impacts of wildfire-generated charcoal and how its presence changes decomposition rates, microbial community composition and function in peatlands is also not well understood. This understanding is important for the establishment of peatland carbon budgets in response to

climate change, contributing to the development of accurate and reliable global carbon budgets and climate modelling that can account for the increasing vulnerability of boreal peatlands to fire.

The overall objective of this study is to improve understanding of peatland CH_4 cycling post-wildfire. Specific objectives are:

1. To quantify CH_4 and CO_2 exchange in a peatland, post-wildfire.
2. To determine the impacts of wildfire and the produced charcoal on the CH_4 production potential and oxidation potential in peatlands, post-wildfire.
3. To determine the impacts of environmental variables such as soil temperature, moisture, organic matter content, phenolic compound concentration, and water table position on CH_4 cycling in peatlands post-wildfire.

1.7 Thesis Structure

This thesis is written in manuscript format and as such some of the information within chapters may have been stated previously. Chapter 2: Wildfire-generated charcoal impacts on CH_4 production and oxidation in a mid-Northern Ontario peatland addresses objectives 2 and 3, including an investigation of the role of phenolic compounds on CH_4 cycling. Chapter 3: Impacts immediately post-wildfire on methane dynamics in a bog peatland in Central Alberta address all three objectives. The concluding chapter includes a summary of the results of both chapters, insights learned from this study, and future work to be considered.

Chapter 2

Wildfire-generated charcoal impacts on CH_4 production and oxidation in a mid-Northern Ontario peatland

2.1 Introduction

Northern peatlands are globally important ecosystems because of their ability to store carbon (C) over long time scales (Clymo, 1987). Northern peatlands store a large amount of soil carbon (~415 ± 150 Gt C; Beaulne et al., 2021), despite only covering 2.8% of global land area (Yu et al., 2010; Xu et al., 2018). When disturbed, peatlands are responsible for C emissions through the production and emissions of greenhouse gases (GHG) such as carbon dioxide (CO_2) and methane (CH_4), which have major implications for climate change. Peat deposits were created in northern regions during the Holocene epoch (Yu et al., 2010), where permanently saturated conditions and cool temperatures sustained the slow accumulation of peat (Clymo, 1987). Undisturbed peatlands function as a C sink as a result of primary production exceeding organic matter decomposition and C loss (Gorham, 1991). Because of their ability to store C, peatlands have contributed to the slowing of global warming, but climate change now poses a significant threat to the stability of peatlands' C stores (Harris et al., 2020). Climate change resulting in changes to ecological conditions (i.e., moisture and temperature) may alter peatland ecohydrological structure and biogeochemical function, impacting the long-term stability of peatland ecosystems (Belyea and Baird, 2006; Waddington et al, 2015; Harris et al., 2020). Impacts on peatland resilience can have severe consequences when peatlands are subject to further environmental change as a result of the increase of severity and extent of natural disasters, such as wildfire (Turetsky et al., 2011).

The combination of reduced resilience, drying, and more frequent wildfires increases the potential for dramatic C storage loss, which would amplify global warming through its impact on global soil C storage (IPCC, 2021). Because of this, understanding the effects of wildfire on C cycling in peatland ecosystems is critical.

Environmental and climatic variables in peatlands, such as annual growing season length and precipitation, temperature, water table position, and vegetation and microbial community composition, affect C cycling in the ecosystem at various temporal and spatial scales (Lund et al., 2010; Marti et al., 2015; Nugent et al., 2018; 2010). It has been well-established in the literature that various environmental variables such as temperature and moisture content have significant impacts on CH_4 dynamics in peatlands (Lai, 2009). Water table position specifically, is an important environmental variable that impacts CH_4 oxidation and production by altering the size of the oxic and anoxic zones (Lai, 2009). Changes in the water table position due to wildfire disturbance can result in either an increase or decrease in CH_4 emissions, depending on how the hydrology of the peatland was altered (Davidson et al., 2019). Water table position also has much larger implications for C storage as the accumulation of organic matter in peatlands is dependent on shallow water tables that reduce decomposition in relation to production (Ingram et al., 2019; Clymo, 1984). Peatlands located in Canada's boreal shield are often formed in bedrock depressions and are prone to varying water table positions as they are dependent on lateral flow from adjacent upland areas (Spence and Woo, 2003). Peat deposits in boreal shield peatlands tend to be shallower than peatlands in the boreal forest due to the location of the underlying bedrock; therefore, they tend to be more prone to drought conditions and have limited water storage capacity

(Dixon et al., 2017). Increases in soil temperature as a result of wildfire can also have implications on C emissions because if the microbial community is still present post-fire, CH_4 emissions would be reduced due to increased oxidation or loss of methanogenic community (Jaatinen, 2004). Warmer soils in peatlands post-wildfire are most likely a result of increased net radiation due to vegetation and tree cover loss during the fire (Lukenbach et al., 2015).

Peatlands are one of the largest natural contributors to global CH_4 emissions, contributing approximately 20% of all global CH_4 emissions (Bridgham et al., 2013). Microbial processes of CH_4 production by methanogenic *Archaea* in anoxic conditions and CH_4 oxidation by methanotrophic *Bacteria* in oxic conditions determine how much CH_4 is emitted from peatland ecosystems (Esson et al., 2016; Marti et al., 2015). Water table fluctuations as a result of a disturbance, resulting in unstable oxic and anoxic zones for methanogens and methanotrophs, can result in a shift of typical CH_4 emissions in a peatland (Davidson et al., 2019; Bridgham et al., 2013). Methanogens rely on access to different sources of labile C that are typically supplied by the presence of vascular plant productivity through root exudates and plant litter (Bridgham et al., 2013). Second to suitable redox conditions, C substrate availability is the most important controlling factor for CH_4 production and oxidation (Ruemer et al., 2018; Ho et al., 2015).

Wildfire produces a charcoal layer on the peatland surface, this layer of charcoal has an adsorptive capacity like activated charcoal and therefore can have various impacts on the microbial community composition and function (Pietikäinen et al., 2000). The charcoal's ability to adsorb organic compounds may result in the formation of new habitat for microbial

communities, which can decompose the adsorbed compounds (Zackrisson et al., 1996). Wildfire generated charcoal in peatlands can enhance C substrate availability as the decomposition of plant litter is sometimes affected by charcoal proximity (Zackrisson and Nilsson, 1996). Multiple studies have also found that microbial biomass is typically increased with the presence of charcoal or biochar in forest and agricultural soils (Zackrisson and Nilsson, 1996; Sam et al., 2017). Multiple studies have shown that charcoal is capable of adsorbing phenolics (Zackrisson et al., 1996; Zackrisson and Nilsson, 1992; Ngoc et al., 2022). However, in peatlands, phenolics are traditionally considered as key biochemical inhibitors of decomposition in anoxic conditions, which has significant impacts on C storage as this process prevents the re-release of C (Freeman et al., 2001; Urbanová & Hájek, 2021). The enzymic 'latch' concept is used to explain the prevention of enzyme phenol oxidase from eliminating phenolic compounds that further inhibit biodegradation by hydrolase enzymes in anoxic conditions in peatlands (Freeman et al., 2001, 2004; Wen et al., 2019). Based on the principles on the enzymic latch concept, the increase in phenolics in a peatland as a result of charcoal produced by wildfire could potentially provide additional suppression of the activity of hydrolase enzymes, inhibiting the mineralization of organic matter at a rate higher than that of a peatland not impacted by wildfire (Freeman et al., 2004; Saraswati, Dunn, Mitsch & Freeman, 2016). This could then limit the substrate supply to methanogens.

This study investigates how position within a peatland, related to microtopography and position relative to the edge of the site affects the response in CH_4 production and oxidation post-wildfire. The role of wildfire-generated charcoal and its impacts on CH_4 production and oxidation is also examined. This work will give new insight into the long-

term impacts of wildfire on peatlands and the implications on C and greenhouse gas exchange.

2.2 Methods

2.2.1 Study Sites

The study area is located 225 km north of Toronto and 160 km south of Sudbury Ontario, in the open rock barrens landscape of the Eastern Georgian Bay Region, which is a part of the Georgian Bay UNESCO biosphere, Mnidoo Gamii. The study sites are situated within the Robinson-Huron Treaty of 1850 and the Williams Treaty of 1923, located on Anishinabek territory. This region is characterized by a hodgepodge of lichen mats, moss cushions, *Sphagnum*-dominated peatlands, forested uplands containing exposed bedrock, moss-dominated ephemeral wetlands, granite bedrock ridges, valleys, and depressions (Wilkinson et al., 2020). Depressions in the granite bedrock support peat accumulation where a layer of mineral soil is overlain by layers of peat and moss. This rock barrens landscape creates a unique hydrogeological setting, which results in hydrological isolation of these small peatlands formed over bedrock. In this region, the 20-year (2002–2020) mean (± s.d.) annual temperature is 6.6 ± 11.3 °C, with mean monthly air temperature in January of -8.5 °C and a mean monthly air temperature in July of 20.5 °C (ECCC, 2021). Long-term annual cumulative precipitation totals 853 ± 251 mm, and the growing season rainfall long-term mean is 452 ± 148 mm.

The burned study site, referred to as “Key River 102” or the “burned” site, was impacted by the Parry Sound #33 wildfire in 2018 where 11,000 ha of this region was

burned. The mean peat depth at Key 102 is 90 cm. The other study site, which is approximately 70 km south of the burned site, in the same region with a similar peat depth and hydrogeologic setting served as an undisturbed reference site as it had not been impacted by the fire. This site is referred to as “Dinner Lake 415” or the “natural” site.

2.2.2 Sample Collection and Study Design

Sample collection took place in August of 2020 where twelve 20 cm deep soil cores were collected from each site. At the burned site, three sets of cores were collected from both the margin and the middle of the peatland to capture a burn severity gradient where depth of burn at the margin was greater. Each set of cores consisted a hummock-hollow pair. This design was replicated at the natural site so that spatial variation under unburned conditions could be evaluated and compared to the burned site. Therefore, at the natural site, six cores were taken from hummocks and another six were taken from hollows, three from the middle of the peatland and three from the margin of the peatland. These cores were divided by depth (0-10 cm and 10-20 cm) and subsampled for use in multiple lab soil incubation experiments and chemical analyses. Soil cores were kept frozen until analyzed.

2.2.3 Potential CH_4 Production

Potential CH_4 production was measured under anoxic conditions using a similar method to Strack et al. (2004) and Davidson et al. (2019). Approximately 15 g of wet peat from each core at each depth was combined with enough distilled water to saturate the sample without allowing for standing water and used to create a slurry of peat. A subset of additional samples from the hollows at each site had wildfire-generated charcoal either added

(for the natural samples) or removed (for the burned samples) in order to examine the potential effect of wildfire-generated charcoal on CH_4 dynamics and microbial activity. Each slurry was incubated in a 250 mL glass jar after being flushed with nitrogen (N_2) for 15 minutes in a glove bag and then sealed to create anoxic conditions for the samples. The peat slurries were incubated at room temperature (between 20–23 °C) and sampled immediately after flushing with N_2, then at 24-hours, 48-hours, and 72-hours. The samples were flushed with N_2 again on day 7 of the incubation and sampled at the same intervals during week 2 of the incubation. Prior to sampling, the peat slurries were manually shaken to adequately combine the gases within the peat pore spaces with the incubation jar headspace.

During sampling, 10 mL of gas was extracted from the peat slurry jars using a needle and syringe punctured through a septa lid so that the sample remained sealed throughout the experiment. The 10 mL gas sample was injected into a flow-through loop attached to a greenhouse gas analyzer (LGR-ICOS GLA132-CCIA2)) and analyzed for CH_4 concentration by comparison to a series of injections of known concentration (1, 5 and 50 ppm). After sampling, 10 mL of N_2 was injected back into the peat slurry jars to maintain headspace pressure. The linear increase (r^2) in headspace CH_4 over the incubation period (0-72 hrs) was used to calculate CH_4 production potential after correcting for dilution of N_2 (Strack et al., 2018). To ensure data quality, r^2 values lower than 0.7 were discarded. CH_4 production potential (mg CH_4 m^{-2} d^{-1}) was calculated according to Equation 1:

$$CH_4 \text{ Production Potential} = \frac{dCH_4}{dt} \times \frac{Vol_{jar} \times MM_{CH_4} \times 15}{MVol \times Dw_{soil} \times 24}$$

where dCH_4/dt is the slope of the measured CH_4 concentration over time during the incubation (μmol mol^{-1} hr^{-1}), Vol_{jar} is the gas volume in the jar (L), MM_{CH4} is the molar mass

of CH_4 (16.04 g mol^{-1}), MV is the molar gas volume at the given temperature, and DW_{soil} is the dry weight of the soil (g), 15 is the weight of the incubated peat sample (g), and 24 is a conversion factors from hours to days.

2.2.4 Potential CH_4 Oxidation

As for potential CH_4 production, potential CH_4 oxidation was measured using a similar method to Strack et al. (2004) and Davidson et al. (2019). However, CH_4 oxidation was measured under oxic conditions. Therefore, the peat slurries were flushed and then sealed using ambient air prior to the incubation. In order to measure the decrease in CH_4 over the incubation period, 10 mL of air was removed and 10 mL of 5ppm CH_4 standard was injected into the jars. To maintain headspace pressure after sampling, 10 mL of ambient air was injected back into the peat slurry jars. The linear decrease (r^2) in headspace CH_4 over the incubation period (0-72 hrs) was used to calculate CH_4 oxidation using Equation 1 after correcting for any addition of CH_4 from the ambient air (Strack et al., 2018). Ambient air samples were injected into the portable greenhouse gas analyzer periodically to measure CH_4 concentration and the daily average of these samples was used for the correction. All other steps outlined in the Potential CH_4 Production section remained the same when measuring and calculating CH_4 oxidation. After the experiment, the jars were dried to determine the dry weight of the peat samples.

2.2.5 Phenolics Analysis

The concentration of phenolics in each of the samples from both sites were analyzed using a similar method to Box (1983). This method uses the Folin-Ciocalteau phenol reagent and sodium carbonate as the supporting medium and is widely used in the assessment of phenolic compounds in peatland soil (Dieleman et al., 2015; Ngoc et al, 2022). Subsamples of all cores from both the 0-10 cm and 10-20 cm depths were analyzed. Each of the samples were homogenized by manual hand mixing while wearing gloves to prevent contamination. Then, 20 g of each homogenized sample was weighed and placed into a centrifuge tube with 20 g of Milli-Q water. The samples were then placed on a rotor and rotated for 24 hours to ensure adequate mixing of the soil and added water. After 24 hours the samples were centrifuged at 1000 rpm and 20 °C (room temperature) for 30 minutes twice, totalling 1 hour. After centrifuging, pore water was extracted from the samples through filtration using a 0.22 µm nylon filter. The extracted pore water was refrigerated for 3–5 days prior to analysis.

For the phenolics analysis, 1 mL of filtered pore water from the samples was pipetted into a smaller centrifuge tube and combined with 0.15 mL of sodium carbonate solution and 0.50 µL of Folin-Ciocalteau phenol reagent. The centrifuge tubes were manually shaken to mix the sample and left to sit for 1 hour and 15 minutes while the chemical reaction took place. Next, 0.30 mL of each sample with the added chemicals was pipetted into a 96 well standard clear assay plate. Three replicates of each sample were used for analysis to account for any variation within the samples. Phenolic dilution standards were also loaded into the assay plate and used to create a calibration curve for phenolic detection. Once full, the clear assay plate was loaded into a Multimode Microplate Reader (manufacturer) and analyzed for

absorbance at 750 nm. The pH of the samples was also measured using a Thermo Fisher Scientific pH probe inserted into leftover filtered pore water that had not been used for phenolics analysis.

2.2.6 Organic Matter

The percentage of organic matter present in each sample was determined using the loss on ignition (LOI) method. Approximately 5 g of oven-dried sample was weighed and placed into a dried clay crucible. Samples were ignited at 550 °C for 2 hours, allowing and additional 1 hour for heat-up time. The resulting ash was weighed after ignition to determine the loss of organic matter. Organic matter was calculated using Equation 2:

$$\% \text{ Organic Matter} = \frac{\text{dry weight (g)} - \text{ash weight (g)}}{\text{dry weight (g)}} * 100\%$$

2.2.7 Data Analysis

All data analysis was completed using the statistical analysis program R (R Core Team, 2013). As the objectives of this study focused on wildfire impacts on CH_4 dynamics, the effect of charcoal treatment, site (natural vs. burned), microform, position on the landscape, depth, and their interaction were included as fixed effects in separate linear mixed effects models for CH_4 production potential and oxidation potential using the package nlme (Pinheiro et al., 2014). Site was included as the random factor for all statistical analyses to account for repeated measures of cores within each site. The effects of charcoal treatment were run during a separate analysis excluding the depth and topography variables as they

were irrelevant to this part of the study. Differences were considered statistically significantly when $p < 0.05$ using the anova output command for each model.

Linear regressions were run to examine relationships between phenolic compound concentrations and production potential and oxidation and a linear mixed effect model was run to determine the effect of site, microform, position, and depth on phenolic concentrations. Linear regressions were also run to examine relationships between organic matter content and production potential and oxidation and a linear mixed effect model was run to determine the effect of site, microform, position, and depth on organic matter.

2.3 Results

2.3.1 CH_4 Oxidation Potential

Mean oxidation potential was highest for samples for hummocks at the natural site, collected from the middle of the peatland at a 10-20 cm depth (0.149 μg C g dry peat $^{-1}$ d^{-1}; Table 2.1). Comparatively, mean oxidation potential was lowest for samples from a hummock at the natural site collected from the margin of the peatland at a 10-20 cm depth (0.046 μg C g dry peat $^{-1}$ d^{-1}). The effects of microform, week, site, position, and depth alone were not statistically significant for explaining variation in oxidation potential. However, the interactions between position and depth ($F_{1, 38}$=4.53, p=0.039) position and site ($F_{1, 38}$=5.74, p=0.021), position, depth, and site ($F_{1, 38}$=11.40, p=0.001) were all significant (Table 2.2). Overall, oxidation potential was slightly higher for burned samples collected from the margin of the peatland and slightly lower for burned samples collected from the middle of the

peatland; however the pattern of oxidation with depth varied (Figure 2.1). At the burned site, deep peat samples had higher CH_4 oxidation than shallow peat samples in the margin, while the opposite pattern was observed at the natural site. In the middle of the peatland, the opposite pattern was observed, with higher oxidation in shallow samples in the burned site with the unburned site had higher CH_4 oxidation potential in deep peat samples. Overall, the range of oxidation potentials for all samples is relatively large and there is no clear difference between the sites.

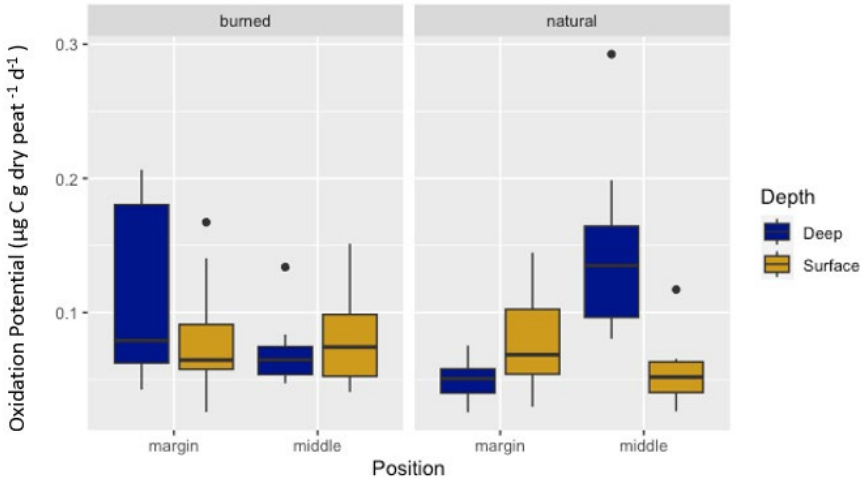


Figure 2.1 Oxidation potentials at each site for the middle and the margin of the peatlands. The incubation weeks were not statistically different from one another ($F_{1,34}$=2.596, p=0.116), so they are grouped together.

Table 2.1 Data Summary Table

Site	Microform	Position	Depth	Mean Oxidation (µg C g dry peat $^{-1}$ d^{-1})	Standard Deviations	Mean Production (µg C g dry peat $^{-1}$ d^{-1})	Standard Deviations	Mean LOI (%)	Standard Deviations	Mean Phenolics (mg/L)	Standard Deviations	Mean pH	Standard Deviations
Burned	Hummock	Middle	Surface	0.080	0.061	0.028	0.010	79.738	0.035	0.623	0.057	6.51	0.054
Burned	Hummock	Middle	Deep	0.059	0.020	0.021	0.002	73.680	0.056	0.689	0.040	6.57	0.064
Burned	Hummock	Margin	Surface	0.123	0.052	0.265	0.604	70.687	0.023	0.703	0.008	6.71	0.063
Burned	Hummock	Margin	Deep	0.134	0.068	0.075	0.073	71.181	0.027	0.681	0.068	6.85	0.040
Burned	Hollow	Middle	Surface	0.098	0.013	0.191	0.418	76.909	0.058	0.704	0.078	4.96	0.043
Burned	Hollow	Middle	Deep	0.072	0.003	0.635	0.861	75.135	0.069	0.713	0.057	4.03	0.042
Burned	Hollow	Margin	Surface	0.061	0.015	0.38	0.556	77.777	0.012	0.603	0.040	6.68	0.054
Burned	Hollow	Margin	Deep	0.141	0.068	0.048	0.001	77.367	0.006	0.698	0.008	6.31	0.052
Natural	Hummock	Middle	Surface	0.051	0.019	0.142	0.303	84.397	0.056	0.385	0.047	5.96	0.483
Natural	Hummock	Middle	Deep	0.149	0.057	0.207	0.382	81.010	0.052	0.285	0.029	4.20	0.584
Natural	Hummock	Margin	Surface	0.091	0.040	0.13	0.262	78.547	0.024	0.386	0.253	4.67	0.674
Natural	Hummock	Margin	Deep	0.046	0.008	0.023	0.003	85.899	0.025	0.388	0.002	6.65	0.897
Natural	Hollow	Middle	Surface	0.063	0.047	0.022	0.002	81.833	0.067	0.375	0.604	5.64	0.784
Natural	Hollow	Middle	Deep	0.103	0.028	1.09	2.322	78.246	0.025	0.492	0.078	6.52	0.889
Natural	Hollow	Margin	Surface	0.0834	0.025	0.134	0.253	78.009	0.026	0.237	0.043	6.56	0.525
Natural	Hollow	Margin	Deep	0.0611	0.019	0.019	0.010	83.015	0.089	0.397	0.042	6.44	0.597

2.3.2 CH_4 Production Potential

CH_4 production potential, site, position, depth, topography and incubation week, and their interactions were not statistically significant for explaining variation among samples (Table 2.2). Mean production potential was highest for samples from hollows at the natural site, collected from the middle of the peatland at a depth of 10-20cm (1.09 µg C g dry peat $^{-1}$ d^{-1}). Comparatively, mean production potential was lowest for samples from hollows at the natural site, collected from the margin of the peatland at a depth of 10-20 cm (0.019 µg C g dry peat $^{-1}$ d^{-1}). Overall, production potentials were similar across all of the samples (Figure 2.2).

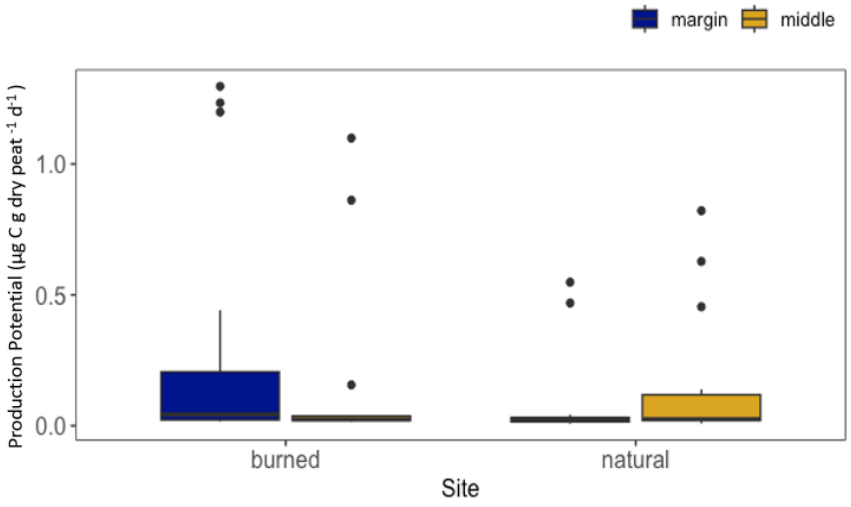


Figure 2.2 Production potentials at each site for the middle and the margin of the peatlands. The incubation weeks were not statistically different from one another ($F_{1,71}$=0.004, p=0.949), so they are grouped together.

	Effect	DF	F-value	p-value
CH_4 Oxidation Potential	Intercept	1,38	-0.731	0.32331
	Position	1,38	0.179	0.674
	Depth	1,38	3.457	0.070
	Microform	1,38	0.603	0.442
	Site	1,38	0.213	0.646
	Position:Depth	1,38	4.527	0.039*
	Position:Microform	1,38	0.058	0.809
	Depth:Microform	1,38	0.555	0.460
	Position:Site	1,38	5.737	0.0216*
	Depth:Site	1,38	0.935	0.33969
	Microform:Site	1,38	1.676	0.203
	Position:Depth:Microform	1,38	0.028	0.866
	Position:Depth:Site	1,38	11.393	0.001**
	Position:Microform:Site	1,38	1.346	0.253
	Depth:Microform:Site	1,38	0.028	0.865
CH_4 Production Potential	Intercept	1,42	1.694	0.942
	Position	1,42	0.458	0.501
	Depth	1,42	0.162	0.688
	Microform	1,42	2.306	0.136
	Site	1,42	0.001	0.967
	Position:Depth	1,42	0.098	0.755
	Position:Microform	1,42	0.217	0.643
	Depth:Microform	1,42	0.741	0.393
	Position:Site	1,42	0.043	0.836
	Depth:Site	1,42	0.716	0.402
	Microform:Site	1,42	0.079	0.779
	Position:Depth:Microform	1,42	0.198	0.792
	Position:Depth:Site	1,42	2.653	0.174
	Position:Microform:Site	1,42	0.039	0.984
	Depth:Microform:Site	1,42	0.002	0.991

Table 2.2 Results from the linear mixed effects models describing effects of position, treatment, site, week, topography, depth, and their interactions on CH_4 production and oxidation potential.

2.3.3 Charcoal Effect on CH_4 Oxidation Potential

Mean oxidation potential was highest in the samples from the burned site where the charcoal was removed (middle= 0.114 µg C g dry peat^{-1} d^{-1}, margin= 0.133 µg C g dry peat^{-1} d^{-1}). Mean oxidation potential was lowest in the samples from the untreated natural samples collected from the middle of the peatland (0.024 µg C g dry peat $^{-1}$ d^{-1}) and the untreated burned samples collected from the margin of the peatland (0.054 µg C g dry peat $^{-1}$ d^{-1}). The effects of charcoal treatment, site, position and incubation week and their interactions were not statistically significant for explaining variation in CH_4 oxidation potential (Table 2.3). Overall, there is no clear charcoal effect on oxidation potential across the various treatments, sites, and sample collection locations (Figure 2.3).

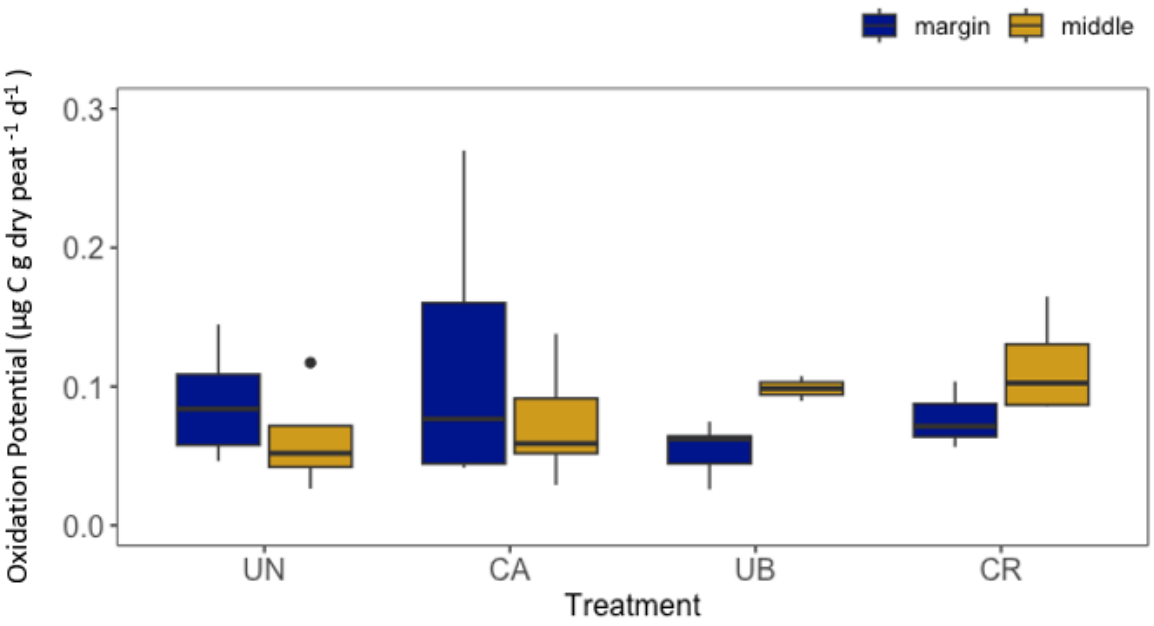


Figure 2.3. Effect of charcoal treatment on CH_4 oxidation during the incubation. UN= untreated natural, CA= charcoal added (to surface of natural hollows), UB= untreated burned, CR= charcoal removed (from surface of burned hollows). The incubation weeks were not statistically different from one another ($F_{1,34}$=2.596, p=0.116), so they are grouped together.

Charcoal Effect Incubation	Effect	DF	F-value	p-value
CH_4 Oxidation Potential	Intercept	7,28	-0.731	0.323
	Position	1,28	1.03	0.32
	Treatment	3,28	1.99	0.15
	Week	1,28	3.26	0.086
	Position:Treatment	3,28	0.92	0.45
	Position:Week	1,28	0.479	0.496
	Treatment:Week	3,28	0.649	0.592
	Position:Treatment:Week	3,28	0.868	0.474
CH_4 Production Potential	Intercept	7,25	0.073	0.942
	Position	1,25	0.676	0.418
	Treatment	3,25	2.97	0.051
	Week	1,25	0.159	0.693
	Position:Treatment	3,25	1.478	0.244
	Position:Week	1,25	0.671	0.420
	Treatment:Week	3,25	3.208	0.04*
	Position:Treatment:Week	1,25	0.292	0.749

Table 2.3 Results from the linear mixed effects models describing effects of position, charcoal treatment, site, and their interactions on CH_4 production and oxidation potential.

2.3.4 Charcoal Effect on CH_4 Production Potential

Mean production potential was highest for the untreated burned samples collected from the margin of the burned peatland (0.53 μg C g dry peat $^{-1}$ d^{-1}) (Figure 2.4). Mean production potential was lowest for the samples collected from both the middle and the

margin of the natural peatland where the charcoal treatment was added (middle=0.036 µg C g dry peat $^{-1}$ d^{-1}), margin= 0.033 µg C g dry peat $^{-1}$ d^{-1}).The effects of charcoal treatment, site, position, and incubation week on oxidation potential were not statistically significant for explaining variation in CH_4 production potential among samples (Table 2.3). However, the interaction between treatment and incubation week was statistically significant ($F_{14,25}$= 3.21, p= 0.04). All other interactions were not statistically significant (Table 2.3). Week 2 production potential was much lower than in week 1, although the treatment effects are relatively similar with the exception of the unburned samples with charcoal added having higher production potential than the other treatments in week 2. But not higher when compared to week 1.

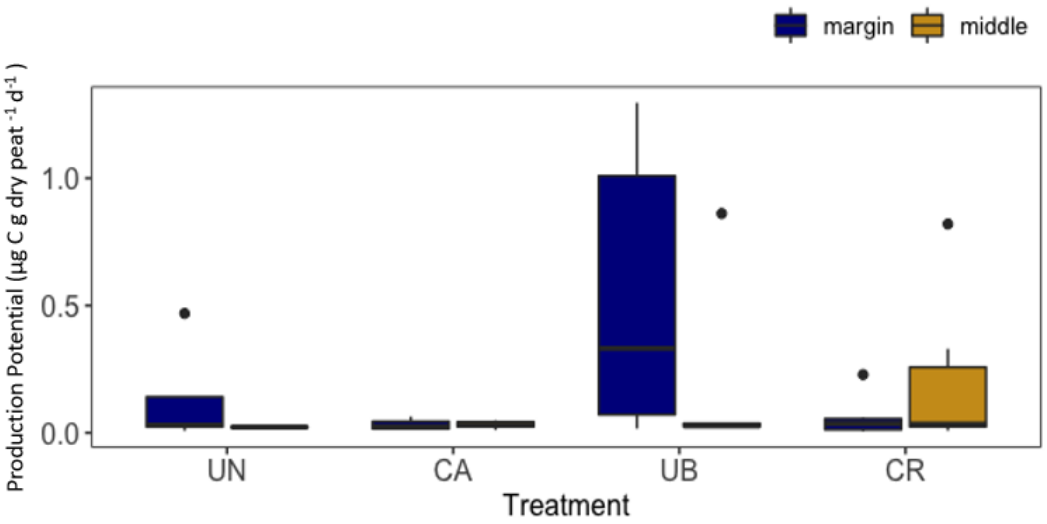


Figure 2.4. Effect of charcoal presence on CH_4 production potential during the first incubation week. UN= untreated natural, CA= charcoal added (to surface of natural hollows), UB= untreated burned, CR= charcoal removed (from surface of burned hollows).

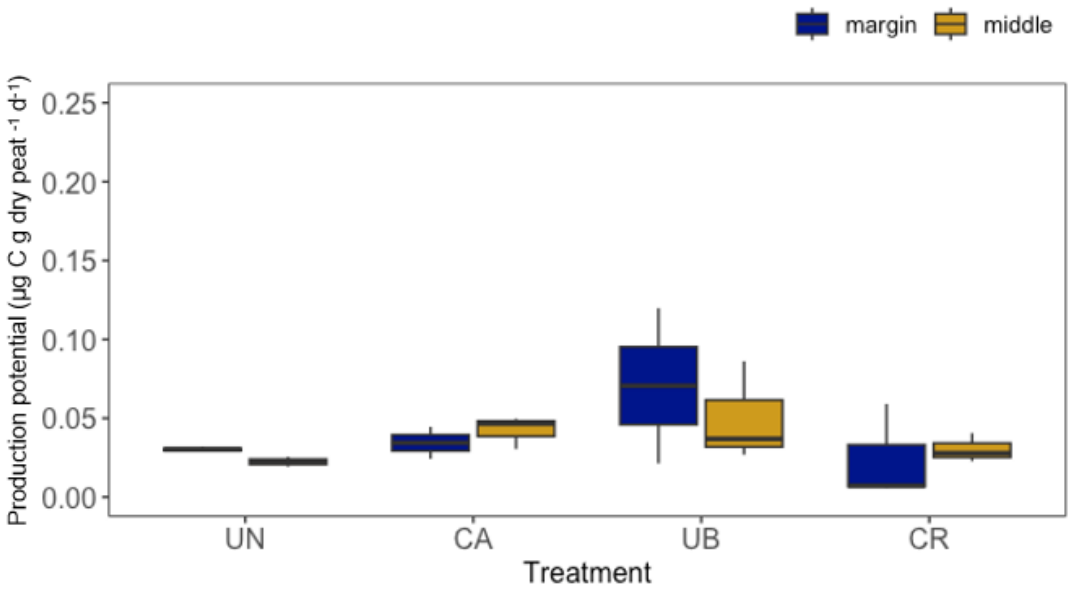


Figure 2.5 Effect of charcoal presence on CH_4 production potential during the second incubation week. UN= untreated natural, CA= charcoal added (to surface of natural hollows), UB= untreated burned, CR= charcoal removed (from surface of burned hollows).

2.3.5 Phenolics

There were consistently higher phenolic concentrations in samples from the burned site compared to samples from the natural site (Figure 2.6) and the relationship between phenolic concentrations and site was statistically significant ($F_{1,46}$=4.47, p=0.038). However, there was no significant correlation between phenolic concentrations and CH_4 oxidation or production potential for any of the samples (Figure 2.7).

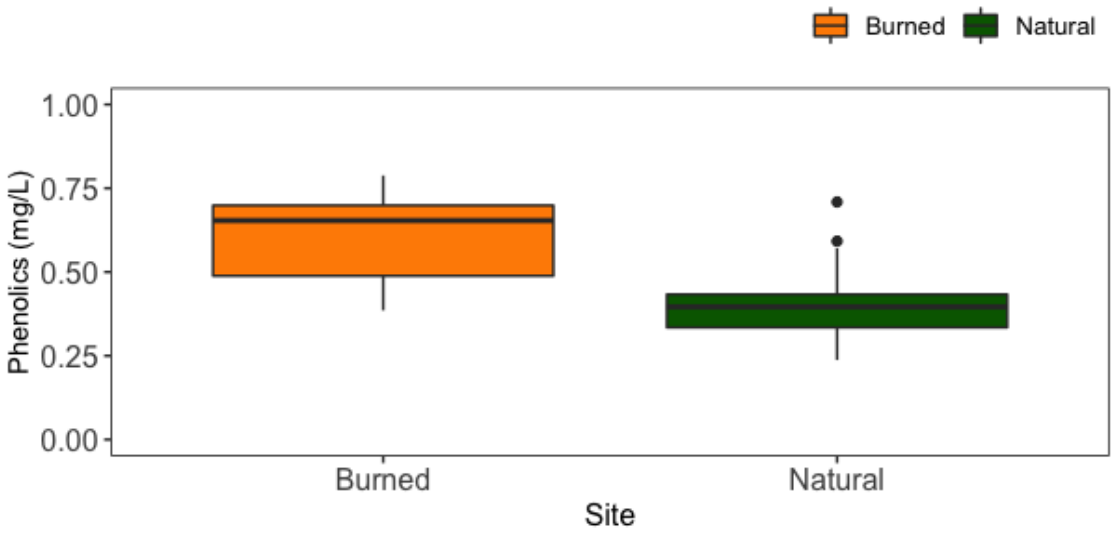


Figure 2.6. Comparison of average phenolic concentrations across all samples and depths from the burned vs. the natural site.

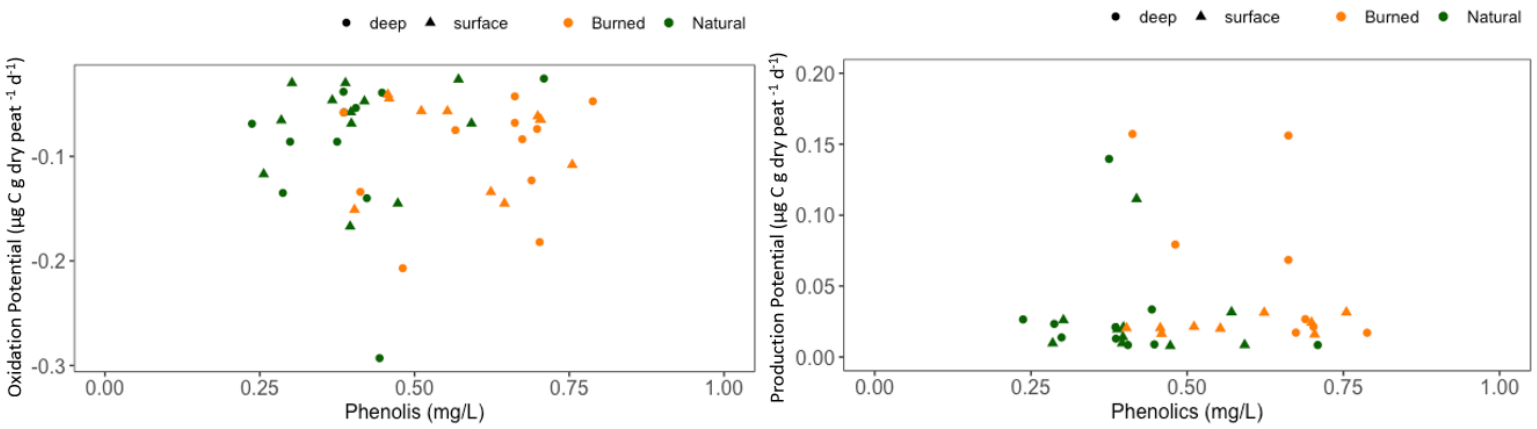


Figure 2.7. Scatterplots revealing no correlation between phenolic concentrations and CH_4 production or oxidation potentials at either site or sampling depths.

2.3.6 Organic Matter Content

Mean organic matter content was highest for samples at the margin of the natural site from hummocks taken from the 10-20cm depth with 85.9% organic matter content. Comparatively, mean organic matter content was lowest for samples at the margin of the

burned site taken from the surface depth (0-10cm) with 70.7% organic matter content (Table 2.1).

2.4 Discussion

2.4.1 Changes in organic matter content suggest low burn severity

Organic matter content at both the burned and natural site were relatively similar, with organic matter being slightly lower at the burned site, although this difference was not statistically significant ($F_{1,46}$=3.83, p=1.788). . While significant soil C loss is to be expected in a peatland post-fire, various factors, such as high fuel moisture and water table levels, can contribute to the preservation of soil C and reduced burn severity (Reddy et al., 2015). The lack of significant differences between the burned and natural site could also be attributed to the time of sampling. Soil samples were collected two years post-fire and while studies suggest that wildfire can disrupt environmental conditions and anaerobic respiration processes, the duration of the persistence of these impacts post-fire is unknown and greatly depend on fire severity and environmental conditions pre-fire (Grey et al., 2021).

Due to the lack of significant difference in organic matter content between the sites, when we would expect organic matter to be much lower at the burned site due to the fire, it can be inferred that either burn severity where the soil cores were collected was low or that ecosystem recovery where the soil cores were collected was significant in the two-years post-fire. Due to the slow recovery of peatlands post-disturbance, it is more reasonable to assume that the burn severity was low at the sample collection locations (Lukenbach et al., 2016). While the study design was created to capture the various topographies and spatial variations

across the landscapes, peatlands have many spatial, temporal, and topographical variations within them which can make it difficult to capture through the collection of select samples (Griffiths et al., 2019).

2.4.2 Methane Cycling: Limited effects of wildfire observed

CH_4 oxidation and production potentials were very similar at both the natural and burned sites, with only the relationships of the interactions between position and depth, depth and site, and the interactions between all three having statistical significance with regards to oxidation potentials. This could be due to many factors again, such as burn severity and time of sample collection. Sample collection occurred two years post-fire and the lack of significant differences in CH_4 cycling at the sites could also be an indication of ecosystem recovery during this period. Alternatively, it could be an indication of a less severe burning at locations where sampling took place, which would explain the similarity between the burned and unburned sites (Wilkinson et al., 2018). Several indicators at the burned site suggest that burn severity was low, specifically the lack of prominent charcoal layer on the collected samples, the lack of difference in organic matter contents between the burned and unburned sites and the presumed ecosystem recovery in the two-years post-fire when samples were collected. This and additional seasonal factors such as high water table heights at the time of the fire could also contribute to the lack of differences between the sites. Despite the lack of difference between sites, when charcoal content was manipulated the addition of charcoal resulted in a reduction of CH_4 production which would be an indicator of the ability of charcoal to reduce CH_4 emissions even with only small amounts present and multiple years

post-fire. Potential controls on shifts in CH_4 cycling post-wildfire are discussed in the following sections.

2.4.3 Phenolic Content

Phenolic compound concentrations were statistically significantly higher in the burned site samples (Figure 2.5), but had no significant relationship with carbon cycling at either site (CH_4 oxidation or production potentials). Increased phenolic content at the burned site was expected as charcoal produced by wildfire was suspected to increase phenolic content in soils at the burned site when compared to the natural. This is due to charcoal's adsorptive capacity of phenolics (Zackrisson et al., 1996; Zackrisson and Nilsson, 1992; Ngoc et al., 2022). With an increase in phenolics we would also expect to see a decrease in CH_4 production potential as phenolics play a key role in the biochemical inhibition of decomposition in anoxic conditions, which has significant impacts on C storage by preventing the re-release of C (Freeman et al., 2001; Urbanová & Hájek, 2021). While there was no direct correlation between increased phenolic content and decreased CH_4 production potential, oxidation potentials were generally higher in the burned samples, which potentially could be related to the higher phenolic concentrations; however, further investigation into mechanisms is needed. Production potentials were very similar at both sites despite differences in phenolic content.

2.4.4 Impacts of Wildfire-Generated Charcoal

When investigating the specific effect of charcoal on CH_4 cycling in an incubation, the interaction between treatment and week were statistically significant when measuring

production potential. It appeared that in the second week of the production potential incubation adding charcoal reduced CH_4 production for the samples from the unburned site, although removing charcoal did not have an effect on the burned samples. The presence of wildfire-generated charcoal was expected to result in a decrease in CH_4 emissions as biochar addition in agricultural soils has a reduction effect of CH_4 emissions as CH_4 uptake is increased (Karhu et al., 2011). The impact of the removal of charcoal from soils is not well studied as it is typically used as a soil addition when looking at the impacts on emissions.

Similar to CH_4 cycling and organic matter content, there were no statistically significant differences between the samples when wildfire-generated charcoal was both added and removed. This was unexpected as the literature suggests that soils amended with charcoal tend to be higher in organic matter due to surface adsorption potential (Zackrisson et al., 1996, DeLuca et al., 2006). The lack of prominent charcoal layer on the samples, most likely caused by sampling two-years post-fire and/or low burn severity where samples were collected could contribute to the lack of difference between the charcoal treatments. Yet, samples from the burned site did have significantly higher phenolic concentrations which would be an indication of charcoal presence, despite lacking a visible prominent charcoal layer. Literature also suggests that the addition of charcoal to soils will result in an increase in soil C sequestration without stimulating CH_4 emissions; this is consistent to what was observed during the second week incubation where the addition of charcoal reduced CH_4 production potential (Koyama et al., 2015). This was consistent with the result that was expected based on findings from other studies where the addition of charcoal (either added through a by-product from wildfire or as an intentional soil amendment) resulted in increased

C storage and decreased CH_4 emissions (Davidson et al., 2019, Koyama et al., 2015, Karhu et al., 2011).

2.5 Conclusion

The results of this study indicate that there were few significant differences concerning CH_4 cycling and organic matter content between samples collected from a burned and unburned peatland with similar peat depth and hydrogeologic setting. This was unexpected as wildfires are considered a major ecological disturbance in peatlands and result in significant changes in environmental variables which are linked to changes in C cycling (Lund et al., 2010; Marti et al., 2015, Harris et al., 2020). The major difference found between the samples was the increase in phenolic content of the burned samples compared to the natural; this was anticipated because of the charcoal by-product of the wildfire. However, the phenolic content increase did not contribute to any dramatic changes in CH_4 cycling, which was not expected as previous studies indicate that an increase in phenolics is associated with a decrease in CH_4 production (Freeman et al., 2004). However, results from the second week of the incubation study that investigates the addition and removal of charcoal reveal that there is a reduction in CH_4 production with the addition of wildfire-generated charcoal, which is consistent with the literature.

Results from this study highlight the need for further investigation regarding the impact of wildfire on C cycling in peatland ecosystems, specifically in the field setting as site specific conditions, including both those related to the site ecohydrology and specific wildfire attributes are likely to drive variation in response.

Chapter 3

Impacts immediately post-wildfire on carbon cycling in a bog peatland in Central Alberta

3.1 Introduction

The province of Alberta has a substantial coverage of boreal peatlands that have been increasingly impacted by wildfires as their severity and frequency increase due to climate change (Kasischke & Turetsky, 2006). Northern peatlands are estimated to store approximately 415 ± 150 Pg carbon (C; Hugelius et al., 2020) and in Alberta they cover over 134,000 km^2 of land (Strack et al., 2019). While peatlands are resilient to many types of natural disturbances, the effects of increased severity and frequency of wildfires on peatlands not well quantified (Nelson et al., 2021). The boreal region is also severely impacted by anthropogenic disturbances such as oil and gas exploration and extraction, and horticultural peat extraction (Rochefort et al., 2022). Therefore, some peatlands in this region may be at risk to both natural and anthropogenic disturbances, which would have increasingly devastating consequences for C storage. A major concern about these disturbances would be the impacts on the production and emission of greenhouse gases (GHG), specifically carbon dioxide (CO_2) and methane (CH_4), which would have serious implications for climate change because they play key roles in the radiative balance of the Earth's atmosphere. Natural, undisturbed peatlands function as CO_2 sinks by slowly accumulating organic matter and act as sources of atmospheric CH_4 (Song et al., 2021).

Environmental and climatic variables such as annual growing season length, precipitation, temperature, water table position, and the vegetation and microbial community in peatlands are controls on C cycling at various temporal and spatial scales (Lund et al., 2010; Marti et al., 2015). Exchanges of CO_2 and CH_4 in peatlands are particularly sensitive to temperature and soil moisture content, where increased temperature will result in an increase in emissions (Song et al., 2020). Temperature and moisture also have a large influence on other environmental variables like microbial activity, biomass, and community composition. Short term variations in temperature and moisture in peatlands can result in significant changes in soil microbial composition, which change typical C storage and release (Lai, 2009, Song et al., 2021). Water table position affects CH_4 oxidation and production by altering the size of the oxic and anoxic zones (Lai, 2009). Changes in the water table position due to wildfire disturbance can result in either an increase or decrease in CH_4 emissions, depending on how the hydrology of the peatland was altered (Davidson et al., 2019).

Wildfire also alters natural peatlands through the introduction of charcoal to the ecosystem, which is generated during fire. The resulting charcoal layer on the peatland surface has an adsorptive capacity similar to activated charcoal and therefore can have various impacts on the microbial community composition and function (Pietikäinen et al., 2000). The adsorption of organic compounds, such as phenolics, may result in the formation of new habitat for microbial communities that can typically decompose the adsorbed compounds (Zackrisson et al., 1996). However, in bog-type peatlands, phenolics are considered as key biochemical inhibitors of decomposition in anoxic conditions, which has

significant impacts on C storage as this process prevents the re-release of C to the atmosphere (Freeman et al., 2001; Urbanová & Hájek, 2021). This process is referred to as the enzymic 'latch' concept, where its principles indicate that an increase in phenolics in a bog-type peatland could potentially provide additional suppression of the activity of hydrolase enzymes, further limiting the mineralization of organic matter (Freeman et al., 2004; Saraswati, Dunn, Mitsch & Freeman, 2016). An increase in phenolics following wildfire could thus reduce rates of organic matter mineralization in the peatland compared to the rate of a peatland not impacted by wildfire and this could then limit the substrate supply to methanogens.

Despite the critical influences both soil moisture and temperature have on C cycling in peatlands, the impacts of wildfire and the resulting significant long and short-term changes to these variables is not well understood. The combination of reduced resilience, drying, and more frequent wildfires impacting peatland ecosystems increases the potential for dramatic C storage loss, which would amplify global warming through its impact on global soil C storage (IPCC, 2021).

This study investigates the effects of changes in environmental variables (moisture and temperature) and soil microbial activity as a result of wildfire on C cycling in peatlands. This was done by measuring in-situ C fluxes and environmental variables almost immediately post-wildfire (2 weeks) in order to examine the implications on C cycling immediately following wildfire disturbance. Soil samples were also collected almost immediately post-wildfire and were incubated and then sampled to determine CH_4 production

and oxidation potentials. The objectives of this study were: (1) to determine the role of wildfire-generated charcoal and its impacts on CH_4 production and oxidation, and (2), investigate the immediate impacts of wildfire on peatland C and greenhouse gas exchange and controlling variables.

3.2 Methods

3.2.1 Study Sites

The study sites are located in Parkland County, Central Alberta. The sites are within Treaty 6 territory and Métis Nation of Alberta Region 4. They are currently on land being leased by the provincial government to Sungro Horticulture. The undisturbed, natural reference site is in the Seba Beach bog and is near an area used for donor material for peatland restoration and ~100 m from active peatland extraction (53°27’45”N, 53°27’45”W). It is classified as a wooded bog peatland but is positioned at the edge of the bog and close to a fen transition zone. The burned site is Tomahawk bog located west of the hamlet, Tomahawk (53°27’45”N, 114°51’07”W). It was impacted during the Tomahawk wildfire that started during the first week of May 2021 and burned 2219.2 ha of land and did not become 100% contained for nearly two weeks. This site is also classified as a wooded bog peatland. The burned site is also in close proximity, ~20 m to a disturbed, recently restored peatland area that was used for peat extraction and is located approximately 20 km northwest of the natural site. These sites are comparable due to similar location, vegetation, hydrology, and peatland type.

3.2.2 Sample Collection, Field Work, and Study Design

Sample collection took place in September 2021 where eight 20 cm deep soil cores were collected from each site. At the natural site four cores were taken from hummocks and another four were taken from hollows, four from the middle of the peatland and four from the margin of the peatland to allow for replication. This method of sample collection was repeated at the burned site. These cores were divided by depth (0–10 cm and 10–20 cm) and subsampled for use in multiple lab soil incubation experiments measuring CH_4 production and oxidation potential. Soil cores were frozen and stored for approximately three weeks before being thawed and prepared for the experiment.

Field GHG fluxes were measured from August to September 2021, with collars installed in the same locations as soil sample collection at each site. Soil sampling took place after fluxes were conducted so that the ground would not be disturbed during flux measurements.

3.2.3 Carbon Dioxide Flux

Carbon dioxide (CO_2) fluxes were measured using the closed chamber method in which a transparent, acrylic cylindrical chamber was placed on a plastic circular collar installed into the peat (Griffis et al., 2000). Eight collars were installed at each of the sites, half were installed on hummocks and half on hollows from the margin to the middle of each of the bogs. The clear chamber with a height of 41 cm and a diameter of 22 cm, was placed inside of a groove of the circular collar that was filled with water prior to each measurement in order to ensure an airtight seal. A battery-operated fan was installed inside of the chamber

to facilitate air circulation. The concentration of CO_2 inside the chamber was measured continuously for three minutes using a portable infrared $CH_4/CO_2/H_2O$ Trace Gas Analyzer (LI-7810, LI-COR, Nebraska, USA). A black tarp was used to create fully dark conditions, enabling ecosystem respiration (ER) to be measured. Order of collar sampled and order of dark vs. light flux was changed daily to account for different light levels and solar angles throughout the day.

Additional environmental factors were measured every time C fluxes were measured; soil moisture was measured with a ML3 ThetaProbe Soil Moisture Sensor (Delta-T Devices) adjacent to the flux collars at 5 cm increments from the soil surface to 25 cm below the soil surface in order to establish a moisture profile. The same method was used to establish a soil temperature profile using a Type-K thermocouple thermometer probe (Omega). Water table depth was measured in a standpipe adjacent to a hummock and hollow at each site and elevation was taken at each standpipe and each fluxing collar so that water table depth could be extrapolated across the site by correcting for the various elevations.

Using the CO_2 fluxes measured under light conditions, the overall exchange and direction of C movement between the atmosphere and the ecosystem, measured under full sun could be determined, also referred to as Net Ecosystem Exchange (NEE). Fluxes measured under dark conditions capture ecosystem respiration (ER), and gross ecosystem productivity (GEP) was calculated as the difference between NEE and ER (Chapin et al. 2006). We used the sign convention where negative values indicate C uptake from the atmosphere and emission to the atmosphere is positive. Raw data was inspected for linearity,

controlling for fit of $R^2 > 0.70$, except for fluxes that were unchanging, representing a flux close to zero. Processing resulted in no data loss.

3.2.4 Methane (CH_4) Flux

CH_4 fluxes were collected using the exact same methodology as the CO_2 fluxes as they were collected simultaneously using a portable infrared $CH_4/CO_2/H_2O$ Trace Gas Analyzer (LI-7810, LI-COR, Nebraska, USA). Raw data was inspected for linearity using the same R^2 cut-off as the CO_2 data, also resulting in no data lost.

3.2.5 Depth of Burn Estimate

To determine the proportion of estimated soil profile that combusted during fire, also known as burn severity, depth of burn (DOB) and peat depth were measured approximately two months post-fire containment. Depth of burn was estimated by establishing transects bound by two unburned reference points, typically exposed tree roots. Transects which used tree roots as a reference were calibrated by adding the average soil depth overlying tree roots measured in the natural reference site. The vertical distance between the reference surface and post-fire surface were measured every 15 cm along the transects, in addition to three soil depth measurements using a peat probe. Estimates of pre-fire soil depth (PFSD) were made by summing the average post-fire soil depth and calibrated DOB at a given transect. The proportion of the profile that burned was determined as the ratio between the DOB and PFSD multiplied by one hundred percent (Wilkenson et al., 2018).

3.2.6 CH_4 Production Potential

Potential CH_4 production was measured under anoxic conditions using a similar method to Strack et al. (2004) and Davidson et al. (2019). Approximately 15 g of wet peat from each core at each depth was combined with enough distilled water to saturate the sample without allowing for standing water to create a slurry of peat. A subset of additional samples from the hollows at each site had wildfire-generated charcoal either added (for the natural samples) or removed (for the burned samples) in order to examine the potential effect of wildfire-generated charcoal on CH_4 dynamics and microbial activity. Each slurry was incubated in a 250 mL glass jar after being flushed with nitrogen (N_2) for 15 minutes and then sealed to create anoxic conditions for the samples. The peat slurries were incubated at room temperature (between 20–23 °C) and sampled immediately after flushing with N_2, at 24-hours, 48-hours, and 72-hours. The samples were flushed with N_2 again on day 7 of the incubation and sampled at the same intervals during week 2 of the incubation. Prior to sampling the peat slurries were manually mixed by shaking the sealed jars to adequately combine the gases in the peat pore spaces with the incubation jar headspace. At the end of the incubation period, jars were dried at 70°C for 48 hours to determine the dry weight of the incubated peat.

During sampling, 10 mL of gas was extracted from the peat slurry jars using a needle and syringe punctured through a septa lid so that the sample remained sealed throughout the experiment. The 10 mL gas sample was injected into a flow-through loop attached to a fast methane analyzer (LGR-ICOS^{TM}, 907-0001-0000-0000) and analyzed for CH_4 concentration

based on a series of injections of known concentration (1, 5 and 50 ppm). After sampling, 10 mL of N_2 was injected back into the peat slurry jars to maintain headspace pressure. The linear increase (R^2) in headspace CH_4 over the incubation period (0–72 hrs) was used to calculate CH_4 production potential after correcting for dilution of N_2 (Strack et al., 2018). To ensure data quality, jars with R^2 values lower than 0.7 were discarded, losing 3.2% of the data. CH_4 production potential (mg CH_4 m^{-2} d^{-1}) was calculated according to Equation 1:

$$\text{Production potential} = \frac{\mathrm{d}CH_4}{\mathrm{d}_t} \times \frac{Vol_{jar} \times MM_{CH_4} \times 15}{MVol \times Dw_{soil} \times 24}$$

where dCH_4/dt is the slope of the measured CH_4 concentration over time during the incubation (µmol mol^{-1} hr^{-1}), Vol_{jar} is the gas volume in the jar (L), MM_{CH4} is the molar mass of CH_4 (16.04 g mol^{-1}), MV is the molar gas volume at the given temperature (L mol^{-1}), and DW_{soil} is the dry weight of the soil (g), 15 is the weight of the incubated peat sample (g), and 24 represents the number of hours per day.

3.2.7 CH_4 Oxidation Potential

As for potential CH_4 production, potential CH_4 oxidation was measured using a similar method to Strack et al. (2004) and Davidson et al. (2019). However, CH_4 oxidation was measured under oxic conditions. Therefore, the peat was incubated at field moisture content and jars were flushed and then sealed using ambient air prior to the incubation. To maintain headspace pressure after sampling, 10 mL of ambient air was injected back into the peat slurry jars. In order to measure the decrease in CH_4 over the incubation period, 10 mL of air was removed and 10 mL of 5ppm CH_4 standard was injected into the jars. The linear decrease (R^2) in headspace CH_4 over the incubation period (0–72 hrs) was used to calculate

CH_4 oxidation after correcting for any addition of CH_4 from the ambient air (Strack et al., 2018). Ambient air samples were injected into a flow-through loop attached to a fast methane analyzer (LGR-ICOS^{TM}, 907-0001-0000-0000) periodically to measure CH_4 concentration and the daily average of these samples was used for the correction. All other steps outlined in the Potential CH_4 Production section remained the same when measuring and calculating CH_4 oxidation.

3.2.8 Organic Matter Content

The percentage of organic matter present in each sample was determined using the loss on ignition (LOI). Approximately 5 g of oven-dried sample was weighed and placed into a dried clay crucible. Samples were ignited at 550 °C for 2 hours, allowing an additional 1 hour for heat-up time. The resulting ash was weighed after ignition to determine the loss of organic matter. Organic matter was calculated using:

$$\%\ Organic\ Matter = \frac{dry\ weight\ (g) - ash\ weight\ (g) * 100\%}{dry\ weight\ (g)}$$

3.2.9 Data Analysis

All data analysis was completed using the statistical analysis program R (R Core Team 2013). As the objectives of this study focused on wildfire impacts on CH_4 cycling, we investigated the effects of site (natural vs. burned), microform, position on the landscape, depth, and their two-way and three-way interactions as fixed effects in a separate linear mixed effects model for both CH_4 production potential and oxidation potential for the

incubations using the R function “lme” (Pinheiro et al., 2014). Site was included as the random factor for all statistical analyses to account for repeated measures of cores within each site. Differences were considered statistically significantly when $p < 0.05$ using the anova output command for each model. Linear regressions were also run to examine relationships between organic matter content, and production and oxidation potential and a linear mixed effect model was run to determine the effect of site, microform, position, and depth on organic matter content.

When investigating the impacts of changes in environmental variables (moisture and temperature) and soil microbial community dynamics as a result of wildfire on C cycling in peatlands using in-situ flux measurements, the effects of site (natural vs. burned), microform, position on the landscape, and their two-way and three-way interactions as fixed effects in a separate linear mixed effects model for both CH_4 flux and CO_2 fluxes (NEE, ER, GEP) was investigated using the R function “lme” (Pinheiro et al., 2014). Plot was used as a random factor to account for repeated measures of the GHG fluxes at each plot over the sampling period.

3.3 Results

3.3.1 Environmental Variables

The summer (June to September) of 2021 in the study region was particularly dry with unusually low precipitation for the study region; the 30-year normal precipitation amount over this period for this region is 59 mm and in 2021 it was less than 10 mm (ECCC, 2021). Therefore, both water table and soil moisture were low at the burned and natural site.

Average water table at both sites was approximately -50 cm below the surface with the exception of the natural hummocks where average water table was much deeper at approximately -70 cm below the surface. There was no relationship between water table position and CH_4 flux at either site or microform (Figure 3.1).

Soil moisture (%) followed a similar pattern where soil moisture was low at both sites; however, it was slightly lower at the burned site for both hummocks and hollows. Average soil moisture at the burned site was 21.5% at the hummocks and 32.0% at the hollows. Average soil moisture at the natural site was 35.7% at the hummocks and 44.6% at the hollows. The relationship between CH_4 flux ($F_{1, 113} = 0.034$, p=0.128), position ($F_{1,113}$ =0.027, p=0.892), site ($F_{1,113}$=2.89, p=0.158), soil moisture ($F_{1,113}$=0.657, p= 0.968) and their interactions ($F_{1,113}$=1.86, p=0.495) was not statistically significant (Figure 3.1).

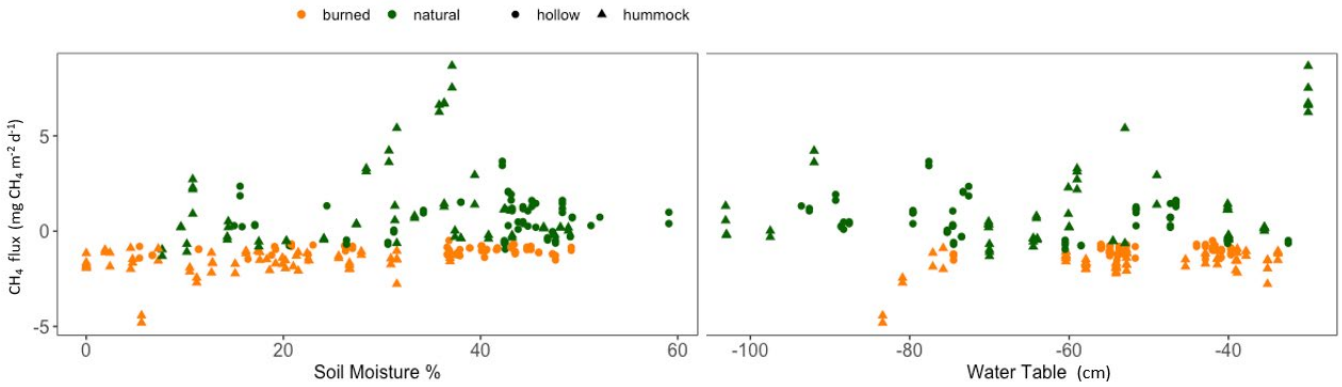


Figure 3.1 Scatterplot showing no correlation between CH_4 flux and soil moisture and water table.

Soils were warmest at the burned hummocks and coolest at the natural hollows. At both sites hummock soil was warmer than hollow soil. However, the differences between soil temperatures at hummocks and hollows at each site were minimal (Table 3.1).

The average depth of burn (DOB) for the margin of the peatland was 0.195 m which was higher than the average DOB for the middle of the peatland which was 0.148 m. The highest DOB measurement was 0.352 m and was taken at the peatland margin (Table 3.1).

Site	Topography	Collar	Mean WT Depth (cm)	Mean Soil Temp (°C)	Mean Soil Mositure (%)	Organic Matter Content (%)	Depth of Burn (m)
Natural	Hummock	1	-71.1	18.7	35.6	80.898	N/A
Natural	Hummock	2	-70.2	17.8	34.9	81.564	N/A
Natural	Hummock	3	-74.5	18.2	37.1	79.543	N/A
Natural	Hummock	4	-68.9	18.9	35.4	83.568	N/A
Natural	Hollow	1	-51.6	17.9	42.3	85.943	N/A
Natural	Hollow	2	-52.3	16.2	45.2	75.937	N/A
Natural	Hollow	3	-56.8	15.8	44.5	76.937	N/A
Natural	Hollow	4	-53.7	16.7	47.6	84.984	N/A
Burned	Hummock	1	-57.8	18.8	21.7	77.893	0.142
Burned	Hummock	2	-62.3	18.4	19.2	81.424	0.149
Burned	Hummock	3	-54.8	18.6	23.8	83.684	0.152
Burned	Hummock	4	-52.9	17.4	21	72.957	0.195
Burned	Hollow	1	-59.1	19.4	34.8	82.924	0.151
Burned	Hollow	2	-58.9	18.9	31.6	70.378	0.148
Burned	Hollow	3	-49.7	17.9	32.9	73.895	0.201
Burned	Hollow	4	-46.6	18.8	32.1	75.895	0.352

Table 3.1 Mean environmental variables.

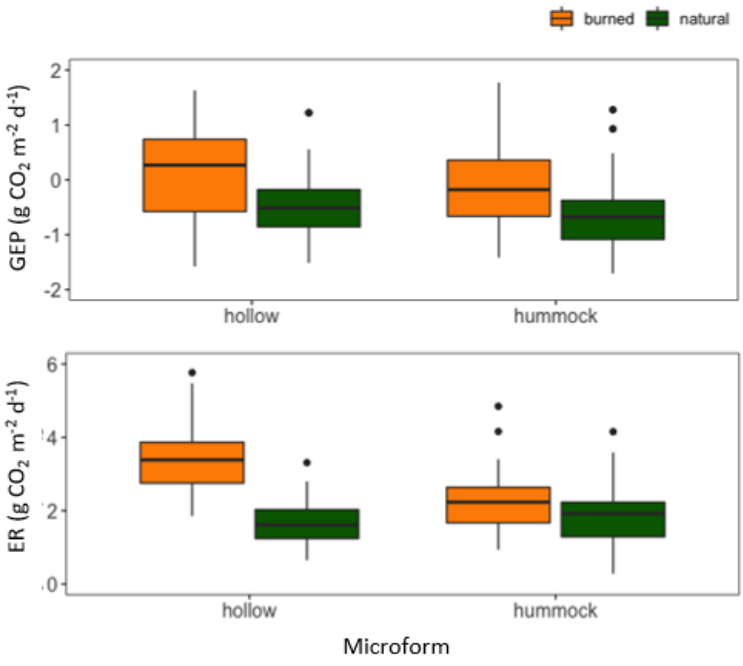
3.3.2 Carbon Exchange

Average values for C fluxes at each site at each plot are provided in the Supplementary Material (Table S1). Ecosystem respiration (ER) was higher at the burned site at both the hollows and the hummocks, than any of the plots at the natural site. Respiration rates were highest at the burned hollows (3.9 g CO_2 m^{-2} d^{-1}) and lowest at the natural hollows (1.7 g CO_2 m^{-2} d^{-1}). The effects of site ($F_{1, 113}$=156.1, $p<0.001$), topography ($F_{1, 113}$=18.9, $p<0.001$, and their interactions ($F_{1, 113}$=53.3, $p<0.001$) were statistically significant for explaining variation in ER. This significant interaction identifies a pattern where in all cases, it appears that the reduction in C uptake/increase in C emissions in response to wildfire is greater at hollows than hummocks.

Average productivity was significantly reduced (i.e., positive values rather than negative) across the flux plots at the burned site and therefore signified a release in C as understory Gross Ecosystem Production (GEP) values were positive representing loss of CO_2 from the system (average burned hollows= 0.556 g CO_2 m^{-2} d^{-1}, average burned hummocks= 0.82 g CO_2 m^{-2} d^{-1}). As GEP should always be negative, these positive values indicate slightly higher measured CO_2 emission under light than dark conditions, possibly due to higher chamber temperatures or photorespiration in the light. Negative GEP values signify C storage and higher productivity, which was present across the flux plots at the natural site (average natural hollows= -0.662 g CO_2 m^{-2} d^{-1}, average natural hummocks= -0.812 g CO_2 m^{-2} d^{-1}). The effects of site ($F_{1, 252}$=23.77, $p<0.001$), topography ($F_{1, 252}$=6.83, $p=0.0095$) and their interactions ($F_{1, 252}$=4.40, $p= 0.037$) were statistically significant for explaining variation GEP.

Overall, NEE was also higher at the burned site than the natural with positive values from the burned site signifying that there was CO_2 being lost from the system. While NEE values for the natural site were also positive, they were much lower than the burned site, across all both microform types. Similarly, to the other flux components, the effects of site ($F_{1, 113}$=98.637, p< 0.001), topography ($F_{1, 113}$=12.941, p<0.001), and their interactions ($F_{1, 113}$= 28.058, p < 0.001) were statistically significant on NEE.

CH_4 fluxes were relatively low overall, but at the natural site were higher than the burned site. In fact, mean measured CH_4 fluxes indicate that the natural system is functioning as a CH_4 source, and the burned site is functioning as a CH_4 sink. Fluxes at both microform types within each site were similar to each other. For CH_4, only the effect of site was statistically significant ($F_{1,252}$=19.2, p<0.001) (Figure 3.2).



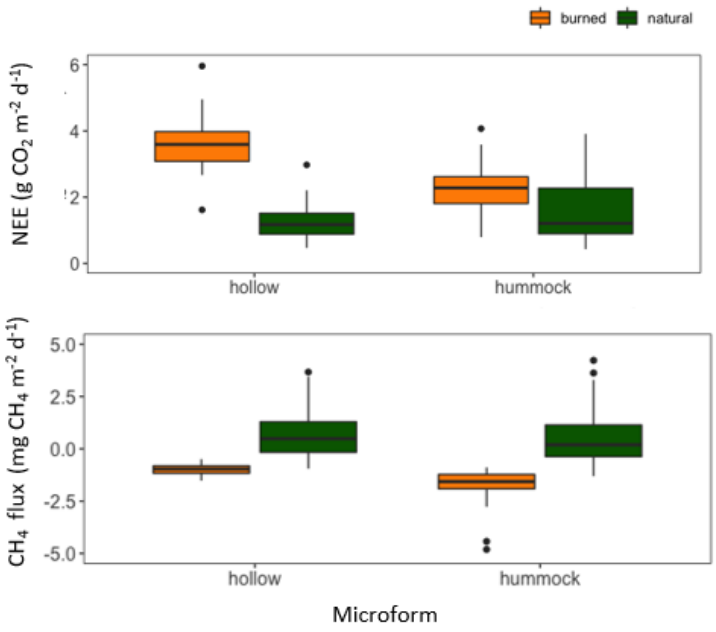


Figure 3.2 Boxplots presenting measured fluxes of CO_2 and CH_4 in a burned and unburned bog. GEP is gross ecosystem production, NEE is net ecosystem exchange and ER is ecosystem respiration

3.3.3 CH_4 Production Potential

The measured CH_4 production potential was average (standard deviation) 0.0016 µg C g dry peat $^{-1}$ d^{-1} at the burned site and 0.0042 µg C g dry peat $^{-1}$ d^{-1} at the unburned site (Figure 3.3). CH_4 production potential was greater at the natural site than the burned site, which is consistent with the field results, and production near the margin of the natural site was higher than the middle; however, site, position, depth, topography and their interactions were not statistically significant for explaining variation in CH_4 production potential (Table 3.2). Week two data was omitted from statistical analysis as there were not a sufficient number of measurements that fit the R^2 quality control cut-off of 0.70.

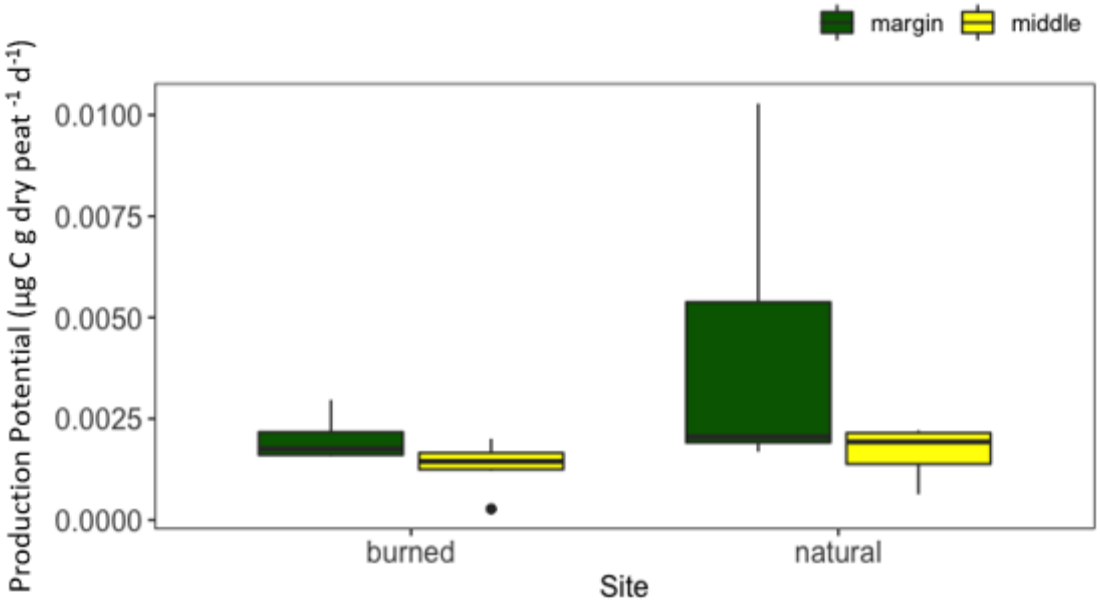


Figure 3.3 CH_4 production potential for samples from the burned vs. the natural site and the margin vs. middle of the sites.

3.3.4 CH_4 Oxidation Potential

The measured CH_4 oxidation potential was average (standard deviation) 0.074 µg C g dry peat $^{-1}$ d^{-1} at the burned site and 0.062 µg C g dry peat $^{-1}$ d^{-1} at the unburned site (Figure 3.4). The incubation weeks were not statistically significantly different from one another, so they have been combined. When measuring oxidation potential, site, depth, and their interactions were not statistically significant (Table 3.2). However, sample position was statistically significant ($F_{1,19}$= 9.2, p=0.0069) with higher oxidation rates in the middle than at the margin. There is an almost significant interaction between site and depth ($F_{1,19}$ = 3.682, p= 0.07), where oxidation potential was greater in samples collected from the surface of the burned site.

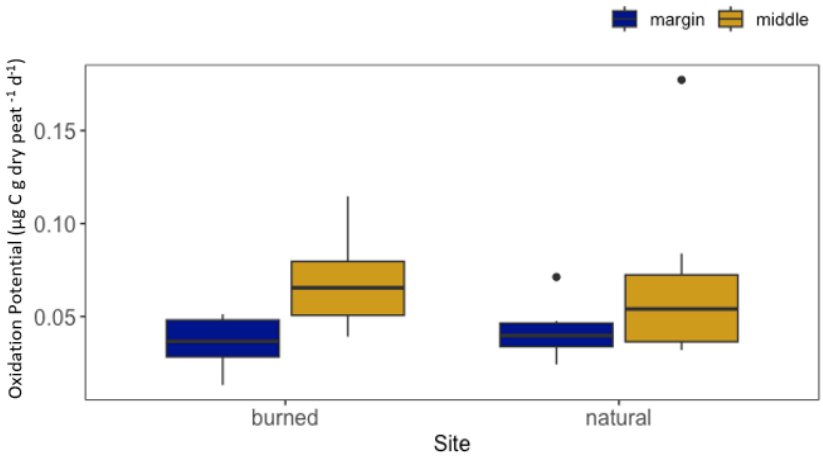


Figure 3.4 CH_4 oxidation potential for samples from the burned vs. the natural site and the margin vs. middle of the sites.

	Effect	DF	F-value	p-value
CH_4 Production Potential	Intercept	1,38	1.326	0.193
	Position	1,38	0.944	0.337
	Depth	1,38	0.024	0.876
	Topography	1,38	0.992	0.325
	Site	1,38	0.001	0.972
	Position:Depth	1,38	0.210	0.648
	Position:Microform	1,38	0.427	0.517
	Depth:Microform	1,38	0.002	0.963
	Position:Site	1,38	0.331	0.568
	Depth:Site	1,38	0.130	0.719
	Microform:Site	1,38	2.406	0.129
CH_4 Oxidation Potential	Intercept	1,19	-1.175	0.254
	Position	1,19	9.1639	0.006**
	Depth	1,19	1.1928	0.288
	Microform	1,19	1.0871	0.310
	Site	1,19	0.0611	0.807
	Position:Depth	1,19	0.4240	0.522
	Position:Microform	1,19	0.2456	0.625
	Depth:Microform	1,19	0.5686	0.460
	Position:Site	1,19	0.3655	0.552
	Depth:Site	1,19	3.6823	0.070
	Microform:Site	1,19	1.1255	0.302

Table 3.2 Results from the linear mixed effects models describing effects of position, site, microform, and their interactions on CH_4 production and oxidation potential.

3.3.5 Effects of Charcoal on CH_4 Production Potential

There was minimal, almost zero CH_4 production occurring during the first incubation week sampling. Therefore, the difference between the incubation weeks were statistically significant ($F_{1,27}$=35.7, $p<0.001$) so they were analyzed separately. During the first incubation week, the measured CH_4 production potential was average (standard deviation) 0.0018 μg C g dry peat $^{-1}$ d^{-1} for the untreated natural samples and 0.0012 μg C g dry peat $^{-1}$ d^{-1} for the natural samples with charcoal added (Figure 3.5). The measured CH_4 production potential was average (standard deviation) 0.0016 μg C g dry peat $^{-1}$ d^{-1} for the untreated burned samples and 0.0014 μg C g dry peat $^{-1}$ d^{-1} for the burned samples with charcoal removed (Figure 3.5). When measuring the impacts of charcoal on production potential, the effects of treatment, site, position, and their interactions on production potential were not statistically significant during the first incubation week (Table 3.1).

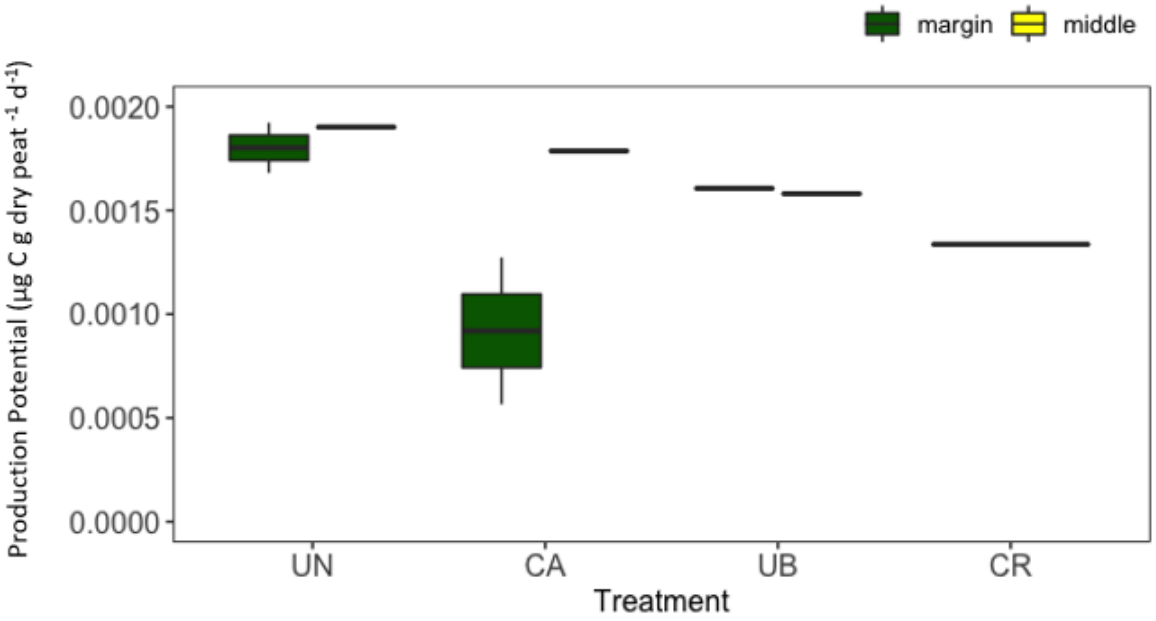


Figure 3.5 Week 1 CH_4 production potential incubations. UN= untreated natural, CA= charcoal added (to surface of natural hollows), UB= untreated burned, CR= charcoal removed (from surface of burned hollows).

During the second incubation week, the measured CH_4 production potential was average (standard deviation) 0.016 µg C g dry peat $^{-1}$ d^{-1} for the untreated natural samples and 0.011 µg C g dry peat $^{-1}$ d^{-1} for the natural samples with charcoal added (Figure 3.6). The measured CH_4 production potential was average (standard deviation) 0.017 µg C g dry peat $^{-1}$ d^{-1} for the untreated burned samples and 0.019 µg C g dry peat $^{-1}$ d^{-1} for the burned samples with charcoal removed (Figure 3.6). During the second incubation week, the effects of charcoal treatment ($F_{4,25}$= 7.3685, p=0.01088), position ($F_{4,25}$= 8.4128, p=0.01988), and site ($F_{4,15}$= 7.3685 p=0.03710) on CH_4 production potential were all statistically significant and

there were no significant interactions (Table 3.1). Overall, the results from week 2 suggest that charcoal presence reduced CH_4 production potential as addition to natural samples reduced production and removal from burned sites increased production.

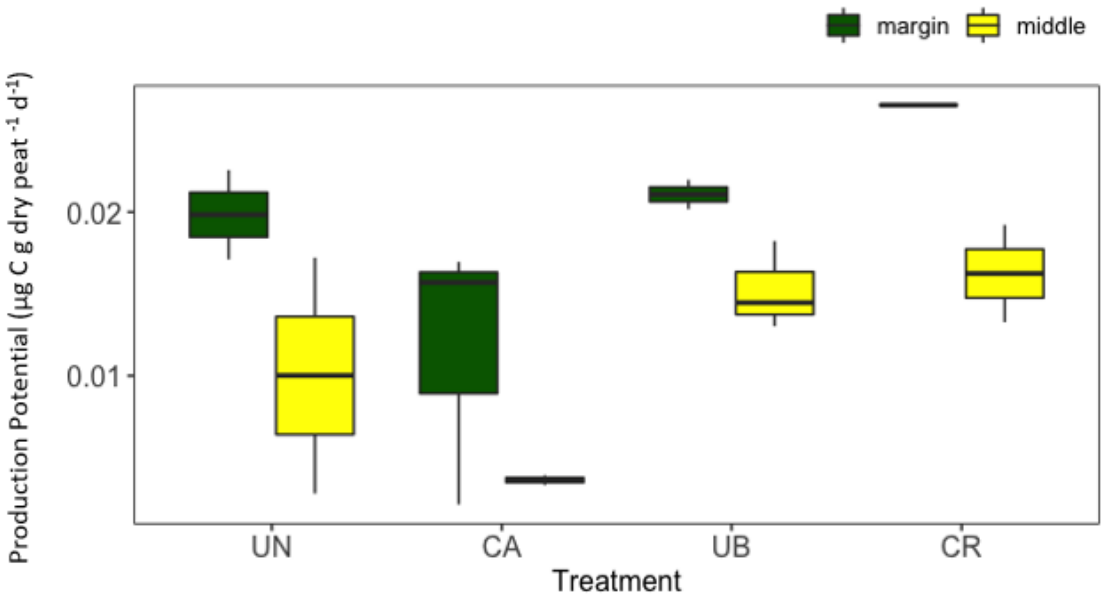


Figure 3.6 Week 2 CH_4 production potential incubations. UN= untreated natural, CA= charcoal added (to surface of natural hollows), UB= untreated burned, CR= charcoal removed (from surface of burned hollows).

3.3.6 Effects of Charcoal on CH_4 Oxidation Potential

Because the incubation weeks when measuring oxidation potential as affected by charcoal presence were not statistically significantly different from one another, they have been analyzed together. The measured CH_4 oxidation potential was average (standard deviation) 0.060 µg C g dry peat $^{-1}$ d^{-1} for the untreated natural samples and 0.056 µg C g dry

peat $^{-1}$ d^{-1} for the natural samples with charcoal added (Figure 3.7). The measured CH_4 oxidation potential was average (standard deviation) 0.074 µg C g dry peat $^{-1}$ d^{-1} for the untreated burned samples and 0.053 µg C g dry peat $^{-1}$ d^{-1} for the burned samples with charcoal removed (Figure 3.7).When evaluating the impacts of charcoal on oxidation potential, the effects of treatment, position, and site, were not statistically significant on their own, but some interactions including position and treatment ($F_{2,25}$=5.82, p=0.032), treatment and site ($F_{4,25}$= 4.54, p=0.046), and the three-way interaction between treatment, position, and site ($F_{2,25}$= 12.49, p=0.005) were all statistically significant predictors of charcoal impacts on oxidation potential (Table 3.2). The interactions reflect that there is somewhat of a pattern where samples without charcoal decreased oxidation potentials and samples with charcoal increased oxidation, but this varies among the margin and middle samples.

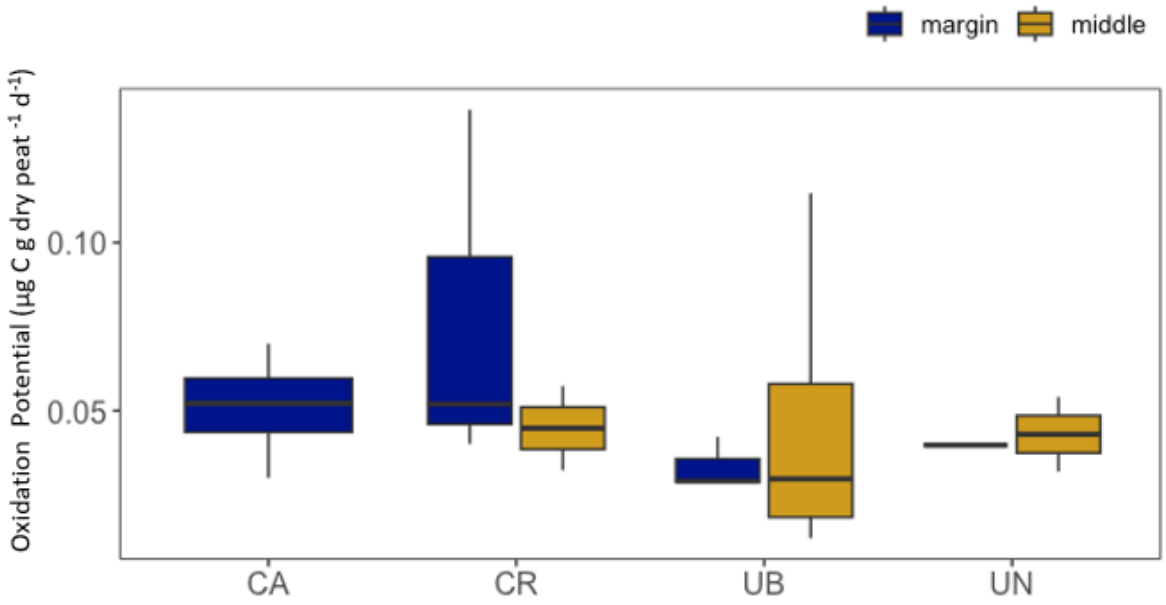


Figure 3.7 Effect of charcoal presence on CH_4 oxidation potential during each incubation week. UN= untreated natural, CA= charcoal added (to surface of natural hollows), UB= untreated burned, CR= charcoal removed (from surface of burned hollows). The incubation weeks were not statistically different from one another ($F_{1,33}$= 0.0896, p=0.7666), so they are grouped together.

Charcoal Effect Incubation	Effect	DF	F-value	p-value
CH_4 Production Potential (Week 1)	Intercept	8,25	3.798	0.012*
	Position	1,25	0.367	0.570
	Treatment	4,25	0.250	0.857
	Site	1,25	0.802	0.445
	Position:Treatment	4,15	0.225	0.806
CH_4 Production Potential (Week 2)	Intercept	8,25	0.492	0.636
	Position	1,25	8.4128	0.019*
	Treatment	4,25	7.3685	0.010*
	Site	1,25	6.2363	0.037*
	Position:Treatment	4,25	0.4660	0.714
CH_4 Oxidation Potential	Intercept	8,25	-2.512	0.040*
	Position	1,25	1.0523	0.339
	Treatment	1,25	1.3577	0.331
	Site	1,25	1.084	0.314
	Position:Treatment	2,25	5.8152	0.032*
	Treatment:Site	4,25	4.5360	0.045*
	Position:Treatment:Site	2,25	12.4871	0.004*

Table 3.2 Results from the linear mixed effects models describing effects of position, charcoal treatment, site, and their interactions on CH_4 production and oxidation potential.

3.4 Discussion

3.4.1 Greenhouse Gas Dynamics in the Field

Both CO_2 and CH_4 fluxes from the understory of the studied wooded bog were relatively low overall, but at the natural site CH_4 fluxes were higher than the burned site, while net CO_2 emissions at the natural site were lower than the burned site (Supplementary Table 1). These values indicate that the ground layer of the natural system is functioning as a CO_2 and CH_4 source. Alternatively, the burned site is acting as a CO_2 source and a CH_4 sink. Inclusion of the overstory primary production at the natural site would like reduce the ecosystem scale CO_2 net emissions (e.g., Murray et al. 2017), potential leading to a CO_2 sink. In contrast, the fire resulted in death of the majority of overstory trees at the burned site suggesting little additional C uptake. Observed differences in C exchange arise due to shifts in both productivity and respiration, alongside changes in methane cycling (see further discussion in 3.4.3 and 3.4.4). Observed changes represent a clear shift in GHG dynamics where there is a reversal in typical peatland emissions. Similar studies that saw this shift in burned peatland ecosystems hypothesized that lower CH_4 emissions at burned peatland sites could be due to factors related to burn intensity which reduces substrate availability, therefore minimizing the methanogenic community reducing CH_4 emissions (Davidson et al., 2019). In addition to this, the decreased CH_4 emissions at both the natural and the burned

sites would be a by-product of the dryness during the sampling period, which resulted in very little CH_4 production but CH_4 oxidation was still taking place.

Understory GEP at the burned site was positive suggesting that respiration under light conditions were slightly higher than respiration in dark conditions, therefore productivity must be zero and this result is an artifact of the calculation where respiration was higher under light than dark conditions. Understory GEP at the natural site was low but negative suggesting that some productivity was taking place. Other studies have investigated the impacts of drought on GEP in forest and peatland ecosystems where GEP would decrease in dry or drought-like conditions (Coursolle et al., 2005). Therefore, the low GEP values at both sites could potentially be explained by the overall lack of precipitation and low water tables during the sampling period at both sites.

Ecosystem respiration (ER) was higher at the burned site overall, which was expected as the fire removed the majority of vegetation resulting in the observed lack of primary productivity and a net CO_2 loss from the system. The effect of site, topography and their interactions significant explained this shift in ER. It appears that the reduction in C uptake and the increase in C emissions in response to wildfire is greatest at the hollows of the burned site. This is most likely due to the increased burn severity at the hollows as they had very little to no vegetation present(Liu et al., 2014).

3.4.2 Environmental Controls on Field C Fluxes

3.4.2.1 Soil Moisture and Water Table

Due to an unseasonably dry summer that impacted both study sites in 2021, soil moisture did not significantly differ between the burned and unburned site or between the microforms. The hollows at both sites were slightly wetter than the hummocks, which was to be expected, especially at the burned site. While water stress is to be expected in Western Canada during late summer, large deviations in precipitation accumulations in the study region relative to the long-term normal were indicated by the Alberta Government as the 0-10 mm received has an estimated frequency of occurring less than once in a 50-year period (Alberta Agriculture and Forestry, 2021). The burned hummocks and hollows were slightly drier than the natural hummocks and hollows, and this dryness would contribute to the lack of CH_4 production at the burned site as these dry conditions are not conducive to CH_4 production (Couwenberg and Fritz, 2012). However, this difference in moisture between the sites would most likely have been more significant if the sampling period experienced a more seasonable level of precipitation. While some studies suggest an immediate wetting of peatland ecosystems post-fire, likely due to the hydrological conditions of the study sites during the study period, this did not take place and the dryness persisted (Lukenbach et al., 2017). Although the relationship between water table and CH_4 flux was not statistically significant, it has been shown that water table position affects CH_4 oxidation and production by altering the size of the oxic and anoxic zones (Lai, 2009). As several studies have reported near zero CH_4 emissions once water table falls below -20 cm (e.g., Couwenberg and Fritz, 2012), it is likely that the lack of CH_4 flux – water table relationship reflects that conditions

at the time of the study were suboptimal for CH_4 production and release at all plot measured, regardless of microform type or wildfire response. In addition to this, the lack of plant substrate as a result of fire would limit CH_4 production and increase CO_2 release as there are no plants for uptake. This increased CO_2 uptake and reduced CH_4 production will likely continue until the vegetation community re-establishes (Dhandapani and Evers, 2020).

3.4.2.2 Soil Temperature

Soil temperatures were highest for the hummocks at the burned site and coolest at the hollows at the natural site, which was expected at each microform post-fire. The burned site overall had slightly higher temperatures than the natural but there were only slight differences between the hummocks and hollows at the burned site. Early on during the sampling period there was potential evidence of smoldering in some areas (i.e., smoking), but not within the sampling area and likely deeper in the peat layer than the soil temperature probe being used. Literature suggests that heat as a by-product of fire will dehydrate and denature organic matter, which would make soil more susceptible to great C losses, most likely from oxidation (Dhandapani and Evers, 2020). Although the relationship between soil temperature and CH_4 flux was not statistically significant, higher soil temperatures would help explain the increased CO_2 and decreased CH_4 emission and as OM content is reduced and remaining OM would be dehydrated post-fire. Soil temperature at the burned site was higher due to lack of plant cover, the presence of charcoal, and a reduction in shade due to reduced tree-cover post-fire, all of which allowed more soil radiation to reach and heat the soil surface.

3.4.3 Wildfire effects on methane production and oxidation potentials

CH_4 production potentials for the burned site measured in the lab were consistent with the field measurements in that they were lower than the natural site. This finding reinforces the reversal in typical peatland CH_4 emissions that is appearing to take place post-fire (Davidson et al., 2019). Because sampling took place immediately post-fire the impacts of wildfire on the ecosystem is dramatic; the recent removal of vegetation and organic matter coupled with an unseasonably dry period results in very dry conditions including a low water table increasing the size of the oxic zone, creating harsh conditions for methanogens and therefore decreasing CH_4 production (Strack et al., 2004; Moore et al., 2011; Davidson et al., 2019). However, even under more ideal hydrological conditions simulated in the lab study, CH_4 production was still low which could be an indication that the microbial community was removed during the fire and that impacts on these microbial communities due to fire could have more broader impacts on the ecosystem's biogeochemical processes (Daniolva et al., 2015). However, potential CH_4 oxidation remained relatively the same in burned and unburned samples which would be an indication of the impact of the dryness of the site in addition to the impact of the charcoal presence on CH_4 production but not oxidation. This would be an indication that the fire did not remove the microbial community and instead the charcoal presence was a key driver of CH_4 production reduction (Sadasivam and Reddy, 2015).

3.4.4 Impacts of Wildfire-Generated Charcoal on CH_4 cycling

Overall, the presence of wildfire generated charcoal appeared to have little impact on CH_4 oxidation while reducing CH_4 production potential. Although, there were many other

factors that would also have contributed to the CH_4 production suppression discussed earlier in this thesis, the addition of wildfire-generated charcoal to the samples from the natural site resulted in a slight reduction of CH_4 production. In contrast, the removal of charcoal from the burned samples did not significantly impact CH_4 production. This could be attributed to the fact that the charcoal removal was done by removing the visible charcoal layer and the biogeochemical characteristics of the soil remained altered as the impacts of charcoal typically persist for longer periods post-fire (Ji et al., 2018; Dhandapani, and Evers, 2020)

With regards to CH_4 oxidation, the interactions between treatment, position, and sampling week were statistically significant. Higher oxidation potentials were observed at the burned samples collected from the middle of the peatland, which would have experienced less severe burning and consistently higher water tables than samples collected from the margin. Overall though, oxidation was relatively not impacted by charcoal presence.

The addition of charcoal (specifically biochar) to soils in order to reduce CH_4 emissions has been studied on agricultural soils on small scales, where the results have been promising in terms of biochar's ability to mitigate carbon emissions (Joseph et al., 2010; Chen et al., 2016; Wang et al., 2020; Chen et al., 2022; Yang et al., 2022). However, some uncertainty remains, particularly surrounding larger scale impacts (Zhang et al., 2022). Research has found it difficult to predict the effects of biochar generated using different materials, which creates different biochar properties and effects also vary among soil types (Wang et al., 2023). The results of this study indicates that CH_4 production reduction does take place charcoal presence.

3.5 Conclusions

Both laboratory incubation and field measured results suggest that there is a reduction in CH_4 emissions in peatlands post-wildfire. While the burned site was no longer emitting CH_4, it was acting a CO_2 source thus having a reversal in GHG dynamics compared to a relatively undisturbed, established peatland. This was expected as post-fire it was hypothesized that a decrease in CH_4 emissions would take place. While there are many different factors as to why potentially we saw very little to no CH_4 flux both in the field and in the lab, this is most likely attributed to the very low water table and lack of precipitation or the severity and recent nature of the burning where the microbial community was no longer active due to the fire. However, the measured continued CH_4 oxidation, both in the laboratory incubation and in the field, would be an indicator of an active microbial community which would instead attribute the reduced CH_4 emissions to charcoal presence through inhibition of CH_4 production.

Sampling the site years post-fire could be beneficial to determine the longer-term effects and recovery and to compare results to similar studies that have mostly been taken with a longer time-period between burning and sampling. Conducting similar measurements and collecting samples during a more typical precipitation period would also be useful in determining the link between wildfire-generated charcoal and suppressed CH_4 emissions without the effect of very low water tables and precipitation reducing CH_4 flux.

Chapter 4

Conclusion

4.1 Summary

The impact of wildfire on greenhouse gas (GHG) dynamics in peatlands tends to vary depending on numerous factors. Factors such as burn severity, time between fire event and sampling, and precipitation/moisture are some of the key differences between the Ontario undisturbed reference and burned site in this study. These factors were not thoroughly investigated in this study, as the focus was GHG dynamics. The impacts of wildfire on GHG dynamics in peatlands are also not fully understood, where research in this thesis begins to fill the gap on how these dynamics can be altered in peatlands post-disturbance. For the Ontario peatlands, where only lab incubations experiments were conducted in which temperature and moisture were controlled, there was little difference between burned and unburned sites when comparing CH_4 oxidation and production potentials. The main difference between the samples was associated with an increase in phenolic content for the burned samples, which is attributed due to the presence of charcoal in these samples as a result of wildfire. However, this increase in phenolic content did not appear to impact the CH_4 cycling in these samples.

Wildfire impacts many environmental conditions in peatlands such as water table position, soil moisture, soil temperature, and plant cover. Changes in these environmental factors in turn have impacts on carbon storage and GHG emissions in peatlands. Water table position affects CH_4 oxidation and production by altering the size of the oxic and anoxic

zones (Lai, 2009). Therefore, a lower water table and a smaller anoxic zone could result in a decrease in phenolic activity with an increase in oxygen and a decrease in phenolic activity is associated with an increase in CH_4 release (Dickopp et al., 2018). In this study for the Alberta peatlands where in-situ measurements were taken, it is more likely that the low water table position and lack of soil moisture is related to an unusually dry period than the wildfire.

The impacts of wildfire on GHG dynamics were more strongly observed when measuring the in-situ C fluxes immediately post-wildfire at the Alberta sites. Through these field flux measurements it was observed that there was a reversal in typical GHG dynamics in peatlands post-fire. These results indicate that post-wildfire, the natural system is functioning as a CH_4 source and CO_2 source in the understory (although likely close to neutral if the overstory was included), and the burned site is functioning as a CH_4 sink and CO_2 source. This reversal of GHG fluxes post-wildfire has been examined in similar studies although not immediately post-fire (Davidson et al., 2019). The lab incubation study using the Alberta samples also support this reversal that was observed in the field where CH_4 oxidation remained relatively steady in burned samples, while CH_4 production declined. The presence of charcoal generated during the fire appears to contribute to the measured reduction in CH_4 production potential.

4.2 Insights

As natural disasters and increased temperatures related to climate change are projected to continue to increase in frequency and magnitude (IPCC, 2021), it is important to understand how wildfire impacts GHG dynamics in major soil C stores, like peatlands. These

insights are useful from both a C accounting/climate projection perspective and from a restoration perspective. Understanding the impacts of wildfire on peatlands will make predicting emissions easier from a local and more global climate perspective, and the results from this study can be used to inform and validate models of peatland C cycling following fire. From a restoration perspective, understanding these dynamics can provide information about the natural recovery process, which can be used for comparison to recovery of ecosystem function post-restoration.

When looking at the results of this study, specifically related to the Alberta sites, an increase in CO_2 emissions and a decrease in CH_4 emissions post-wildfire in peatlands could be considered positive if only looking at reducing GHG emissions as CO_2 is a far less potent greenhouse gas (Chambers, 2003). However, as peatlands are complex ecosystems it would be important to look at restoration from a more holistic perspective.

4.3 Future Research

As this study is one of the first to investigate the impacts on GHG dynamics immediately post-fire in peatlands, using both in-situ field measurements and lab incubation experiments, further study is required. This should include application and study of both in-situ measurements and lab experiments at different time intervals post-fire and across a range of both burned and undisturbed peatlands with different hydrologic and climate settings and dominant vegetation types. Burn severity of the wildfire-impacted peatlands (i.e., depth of burn, fire length, presence of smoldering, etc.) should also be considered in order to account for the variation of wildfires that take place in peatlands.

Finally, to help determine the potential drivers of changes in GHG dynamics in peatlands post-wildfire it would be important to understand the soil microbial community dynamics. Measures of composition, structure and function of the microbial community should be compared between the undisturbed and burned peatlands to get a better sense of the drivers behind changes in carbon cycling and the recovery of the microbial community post-fire. As this study did not include soil microbial community dynamics it can not determine whether environmental factors post-wildfire resulted in a change in how the microbial community functions or if the microbial community was wiped out completely post-fire (Esson et al., 2016).

The lack of this reversal in GHG dynamics for the burned Ontario site could potentially represent ecosystem recovery taking place 2 years post-fire, although this can be difficult to determine without conducting in-situ fluxes. Therefore, taking measurements and collecting samples at different time periods in relation to wildfire would also be important to consider in future study.

Cumulatively, having a greater understanding of the impacts of wildfire on GHG dynamics in peatlands will lead to more insight when making decisions regarding conservation efforts and climate change projections. This is specifically important for Canada where one quarter of the world's peatlands are found (Tarnocai, Kettles & Lacelle, 2011).

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Appendices

Appendix A

Site	Topography	Collar	NEE ($g\ CO_2\ m^{-2}\ d^{-1}$)	ER ($g\ CO_2\ m^{-2}\ d^{-1}$)	GEP ($g\ CO_2\ m^{-2}\ d^{-1}$)	Methane flux ($mg\ CH_4\ m^{-2}\ d^{-1}$)
Natural	Hummock	1	2.4	1.8	-0.6	-0.4
Natural	Hummock	2	2.4	2.2	-0.6	-0.4
Natural	Hummock	3	2.5	1.8	-0.3	-0.4
Natural	Hummock	4	2.4	1.8	-0.2	-0.3
Natural	Hollow	1	1.2	2.7	-0.8	-0.3
Natural	Hollow	2	3.6	1.0	-0.9	-0.3
Natural	Hollow	3	2.4	1.4	-0.7	-0.3
Natural	Hollow	4	3.0	1.4	-1.7	-0.3
Burned	Hummock	1	4.1	4.1	0.9	-4.8
Burned	Hummock	2	6.4	5.4	0.9	-4.4
Burned	Hummock	3	4.0	5.4	0.4	-2.7
Burned	Hummock	4	5.5	3.6	0.5	-2.7
Burned	Hollow	1	5.8	3.7	-0.8	-2.4
Burned	Hollow	2	4.3	4.3	0.8	-2.2
Burned	Hollow	3	5.8	3.1	-0.6	-2.1
Burned	Hollow	4	5.9	4.4	0.5	-2.1

Table S1. Average field flux values measured at the Alberta study sites reported in Chapter 3

Appendix B

Figure S2. Image of the burned Alberta site

Figure S3. Image of the burned Ontario site