

Comparing wildfire recovery at a bog and a fen along a burn severity gradient

by

Emma Janel Ocean Wegener

A thesis

presented to the University of Waterloo

in fulfillment of the

thesis requirement for the degree of

Master of Science

in

Geography

Waterloo, Ontario, Canada, 2024

© Emma Janel Ocean Wegener 2024

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.

I understand that my thesis may be made electronically available to the public.

Abstract

Wildfires are increasing in intensity and severity, emitting carbon (C) stored in soil and biomass to the atmosphere. This is of increased importance in peatlands which have deep deposits of combustible soil. Carbon in peatlands has been accumulating for millennia, due to organic matter input exceeding decomposition and combustion. The presence of saturated soils inhibits the rapid oxidation of dead organic matter, thereby limiting C losses through decomposition. Carbon accumulation in peatlands is supported by the adaptations of characteristic vegetation assemblages, which can grow quickly and in high abundance, increasing the rate of C accumulation, or reducing rates of decay. As wildfires are becoming increasingly severe, collecting data over a range of burn severities, in an array of peatland types to better characterize rates of recovery is paramount. Thus, I measured C fluxes, plant functional traits, and plant community composition at a bog and a fen along a burn severity gradient, with the aim of gaining a better understanding of the influence of burn severity on recovery in different peatland types. I found that, 6 to 7 years following wildfire, biomass accumulation was greater at the fen than the bog, especially the moderately burned fen with nearly 12-fold the biomass of the moderately burned bog; however, the plant community composition was dominated by opportunistic plants such as *Betula glandulosa* that were not characteristic of the unburned treatment. Plant functional traits suggested that response to disturbance differs among plant types along the burn severity gradient at each peatland type, where LDMC is regularly decreased along the burn severity gradient, and either SLA or height are increased. Understory GEP and ER are significantly greater at the fen than the bog, although NEE was not statistically different, as sequestration and efflux balance at each site to approximately $0 \text{ g C m}^{-2} \text{ day}^{-1}$. These results should be considered alongside tree and vegetation surveys, which suggest that while ground level fluxes may be similar, overstory contributions to each site are crucial to consider as they contribute to the carbon storage capacity of the unburned sites, but are missed by ground-level fluxes. There is a great amount of C held in trees at the unburned sites, less at the moderately burned sites due to competition, and a high number of tree seedlings at the severely burned sites. CH_4 fluxes, however, appear to recover more slowly following deeper peat combustion, with moderate burns trending toward pre-burned conditions, while the severe burns have limited efflux of CH_4 , which may suggest that there has been a reduction in substrate quality or that the soil microbial community that has not yet recovered. The results of this study emphasize the need for more nuanced consideration of

burn severity in peatland management and research. Severe burns are becoming more common with climate change, and implementation of burn severity into global C models is necessary to ensure accurate estimates of C losses from wildfire. This study also highlights the importance of distinguishing between bogs and fens in ecological modeling, as applying the same rates of C efflux or accumulation could lead to significant inaccuracies. Understanding these differences is crucial for prediction of C dynamics in peatland ecosystems, particularly in the context of increasing wildfire frequency and intensity.

Acknowledgements

This project would not have been possible without the help and support of many individuals in the lab and in the field. I would like to thank Sara Claussen, Justin Yu, Veronica Santia, and Daniel Quieroz De-Sousa for countless hours spent on data collection. I would also like to thank Kayla Linden, Matthew Morrison, Nataša Popović, and Nicole Balliston, Fieldwork builds complex relationships, where you work with the people you live with, who are additionally the only people you know in a town far from home. I thank you all so much for our time spent building a family, we've had beach days, and carnival trips, but you've also supported me through the frustrations and upsets that come with the bizarre situations surrounding fieldwork. Thank you.

I would also like to thank Dr. Maria Strack for her mentorship and support throughout the collection and writing processes. The opportunities you have provided me have allowed me to build confidence in myself as a scientist, leader, and student.

Dr. Owen Sutton was paramount in the latter stages of this thesis, and I would like to thank him for his time he spent sending back and forth countless edits. Not only was he supportive in the writing process; he has supported me through the trials of life inside and outside of graduate school. I am eternally grateful for his patience, confidence, love, and support.

To my parents, Chris and Tracy, and brothers, Lynden, Jalen and Keiffer – thank you for supporting me through my irrational ambitions, never questioning my capabilities, and lending your own shoulders when the weight was too great for mine. I love you all.

I acknowledge that I live and work on the traditional unceded lands of the Haudenosaunee, Anishinaabe, and Neutral People, and that my research was conducted on the traditional lands of the Cree and Dene Peoples. I respect the sovereignty, lands, histories, languages, knowledge systems, and cultures of First Nations, Métis and Inuit nations.

Funding was provided by NSERC Collaborative Research and Development Grant (CRD) with direct funding from Suncor Energy Inc., Imperial Oil Resources Limited and Teck Resources Limited.

Dedication

I dedicate this work to my parents, Christopher and Tracy Wegener. Thank you for your constant support and love. You have provided the stable ground on which I've been able to flourish as a woman and as a scientist. Nothing I have accomplished has been alone, I owe it all to you.

Table of Contents

Author's Declaration	ii
Abstract	iii
Acknowledgements	v
Dedication	vi
List of Figures	ix
List of Tables.....	xi
Chapter 1	1
Literature review	1
1.1 Introduction	1
1.2 Peatland composition and importance.....	2
1.3 Understanding carbon transformation in peatland ecosystems	2
1.4 Assessing varied responses to burn severity.....	6
1.5 Post-fire dynamics: environmental responses and plant community adaptations	8
1.6 Changes in carbon cycling to post-fire conditions	12
1.7 Impacts of climate change - altered dynamics and increasing fragility.....	14
1.8 Remaining questions and knowledge gaps in peatland wildfire research	16
Chapter 2	21
Vegetation composition and function in peatlands post-fire: plant community composition and plant functional traits.....	21
2.1 Introduction	21
2.2 Methods.....	25
2.2.1 Site location and description.....	25
2.2.2 Vegetation surveys	28
2.2.3 Biomass	29
2.2.4 Plant functional traits.....	31
2.2.5 Data analysis.....	33
2.3 Results	34
2.3.1 Vegetation community composition.....	34
2.3.2 Plant functional traits.....	37
2.4 Discussion	45

2.4.1 Vegetation community composition.....	45
2.4.2 Plant functional traits.....	48
2.5 Conclusion.....	50
Chapter 3	52
Post-fire CO ₂ and CH ₄ fluxes as an indicator of recovery in boreal peatlands.....	52
3.1 Introduction	52
3.2 Methods	55
3.2.1 Site location and Description.....	55
3.2.2 CO ₂ flux.....	56
3.2.3 CH ₄ flux.....	57
3.2.4 Collar vegetation composition and greenness	58
3.2.5 Environmental variables.....	59
3.2.6 Data analysis.....	60
3.3 Results	61
3.3.1 Environmental variables.....	61
3.3.2 Plant community composition and greenness	68
3.3.3 Carbon gas exchange.....	71
3.4 Discussion	90
3.4.1 CO ₂	90
3.4.2 CH ₄	98
3.5 Conclusion.....	100
Chapter 4	101
Conclusion.....	101
4.1 Plant community composition, biomass, and plant functional traits	101
4.2 Carbon cycling	105
4.3 Implications of the results for modelling post-fire carbon cycling in peatlands	107
4.4 Future research needs	108
References	110
Appendix	120

List of Figures

Figure 1: Site map	27
Figure 2: Site photographs.....	28
Figure 3: Examples of the photograph process for measuring plant functional traits	32
Figure 4: Bar graph depicting the variation in total aboveground biomass across all six sites	36
Figure 5: Bar graph depicting the variation in total aboveground biomass across all six sites	37
Figure 6: Boxplot depicting the variation in height (cm) across the different sites.....	39
Figure 7: Boxplot depicting the variation in specific leaf area (SLA, cm^2/g) across the different sites	40
Figure 8: Boxplot depicting the variation in leaf dry matter content (LDMC, g) across the different sites.....	41
Figure 9: Bar graph depicting height (cm), specific leaf area (SLA, cm^2/g), and leaf dry matter content (LDMC, g) of <i>R. groenlandicum</i> across the different sites	42
Figure 10: Bar graph depicting height (cm), specific leaf area (SLA, cm^2/g), and leaf dry matter content (LDMC, g) of <i>P. mariana</i> across the different sites	43
Figure 11: Fort McMurray climographs for 2022, 2023, and 30-year average climate data measured at the Fort McMurray International Airport	62
Figure 12: Boxplot depicting the variation in water table levels across different sites	63
Figure 13: Boxplot depicting the variation in near surface soil temperature (5 cm) across different sites.....	65
Figure 14: Boxplot depicting the variation in 30 cm soil temperatures across different sites.....	67
Figure 15: Boxplot depicting the variation in Green Chromatic Coordinate (GCC) across different sites.....	69
Figure 16: Scatterplot depicting the relationship between gross ecosystem productivity (GEP), and green chromatic coordinate (GCC)	70
Figure 17: Bar graph depicting the average percent cover measured from all eight collars at each site	71
Figure 18: Boxplot depicting the variation in Gross Ecosystem Productivity (GEP) across different sites.....	73
Figure 19: Scatterplot depicting the relationship between Gross Ecosystem Productivity (GEP) and water table at each site.....	74
Figure 20: Scatterplot depicting the relationship between Gross Ecosystem Productivity (GEP) and 5 cm soil temperature at each site.....	75
Figure 21: Boxplot depicting the variation in Ecosystem Respiration (ER) across different sites	77
Figure 22: Scatterplot depicting the variation in Ecosystem Respiration (ER) across water table depths (cm)	79
Figure 23: Scatterplot depicting the variation in Ecosystem Respiration (ER) across 5 cm soil temperatures ($^{\circ}\text{C}$)	80

Figure 24: Boxplot depicting the variation in Net Ecosystem Exchange (NEE) across different sites	82
Figure 25: Boxplot depicting CH ₄ flux across different sites. The y-axis represents CH ₄ , while the x-axis shows the different sites sampled.....	84
Figure 26: Scatterplot depicting the variation in CH ₄ fluxes across different water table depths (cm)	85
Figure 27: Scatterplot depicting the variation in log scale transformed CH ₄ fluxes across different water table depths (cm)	86
Figure 28: Scatterplot depicting the variation in CH ₄ fluxes across different soil temperatures at 30 cm	87

List of Tables

Table 1: Allometric equations for conversion of diameter at breast height (DBH, 1.3) to grams (g)..	31
Table 2: Main effects of plant functional traits and biomass compared across sites.....	44
Table 3: Main effect of site on percent cover of each functional group measured.....	45
Table 4: Diversity indices for understory plant community along bog and fen burn severity gradients	45
Table 5: pH and EC values from each site	59
Table 6: Main effects for CO ₂ , CH ₄ , GCC, and environmental variables	87
Table 7: Main effects of collar vegetation surveys.....	90

Chapter 1

Literature review

1.1 Introduction

Wildfires are happening with greater frequency and severity than ever before in the boreal region due to a changing climate (Turetsky et al. 2015), burning stored organic matter and releasing its carbon (C) into the atmosphere. This is increasingly important in peatland ecosystems, which are becoming more vulnerable to burning, and where C has been accumulating for millennia as peat (Yu et al. 2010). Peatlands are exceptionally effective at storing C due to a typically high water table which inhibits rapid oxidation of dead organic matter, thereby limiting C losses through decomposition. The C accumulation function of peatlands is intimately related to the unique adaptations of characteristic vegetation assemblages. Peatland vegetation communities can grow quickly and in high abundance, increasing the rate of C accumulation as in the case of sedges, or contributing to reduced rates of decay as in the case of Sphagnum mosses, which alter the environment through a host of complex ecological feedbacks (Waddington et al. 2015).

The release of stored C due to wildfire contributes to global climate change, which particularly affects northern peatlands through increased drought and soil temperatures (Li et al. 2017; Wilkinson et al. 2023), making them more susceptible to combustion, thus creating a positive feedback. Although the importance of this disturbance is known, peatland fires have not yet been successfully included in global C cycle models (Bowman et al. 2009; Bona et al. 2020; Nelson et al. 2021), underrepresenting the true efflux of C from the terrestrial pool to the atmosphere. Quantitative data are needed to better understand the impact of peatland fires on the global C cycle, as well as wildfire return intervals based on post-fire conditions (Wilkinson et al. 2023). This study aims to quantify recovery of C exchange and plant communities in peatlands affected by wildfire.

1.2 Peatland composition and importance

While peatlands cover only 2–3% of the Earth’s surface, they store approximately 25–33% of global soil C (Gorham 1991; Turetsky et al. 2014; Van Der Werf et al. 2017). The C stored in peatlands accounts for an estimated 500–600 Gt and is comparable to the amount of C stored in the pre-industrial atmosphere (Turetsky et al. 2014; Intergovernmental Panel On Climate Change 2023). In Canada alone, northern peatlands occupy only 12% of the total land area but store 50% of its terrestrial C stock (Tarnocai 2006). High-latitude peatlands around the globe have accumulated an estimated 547 Gt C since the last glacial maximum, and currently store upwards of 80% of the C in peatlands globally (Yu et al. 2010). During the Holocene, it is thought that the climate would have been significantly wetter than current climatic conditions, contributing to rapid peat expansion (Loisel et al. 2014), with peatlands capable of sequestering over 5 Gt C per century (Yu et al. 2010). In contrast, conditions during the Anthropocene are trending toward being warmer and drier than the conditions that allowed for peat accumulation in the Holocene and are approaching a critical point in which peatlands may begin to shift from sinks to sources, especially in combination with disturbance events (Wilkinson et al. 2023; Zhao and Zhuang 2023).

1.3 Understanding carbon transformation in peatland ecosystems

Peatland C cycling is driven by the balance between C inputs via plant productivity and lateral influx with water, and C outputs including CO₂, CH₄, waterborne losses, and combustion. Carbon enters the peat complex as dead organic matter (Bubier et al. 1993), or as dissolved organic C through ground and surface water, entering the system from upgradient, and leaving the system downgradient (Strack et al. 2008). Carbon can also be lost through the release of greenhouse gases (GHGs) with decomposition.

Although the rate of decomposition is slowed in saturated soils, C losses still occur through trace gas emissions, but these losses are often outweighed by the new addition of dead organic matter (Clymo 1984). Average rates of peat accumulation are 24–32 g C m⁻² yr⁻¹ over the past millennium, with a long-term time-weighted mean rate of 22.9 ± 2.0 g C m⁻² yr⁻¹ (Loisel *et al.*, 2014).

A shallow water table depth is crucial in regulating decomposition within peatlands. Carbon dioxide, which is the main byproduct of organic matter decomposition, is produced predominantly in the oxic layer (Bubier *et al.* 1993), which is the portion of peat above the water table that has some degree of aeration. Thus, the water table position regulates the presence of oxygen, affecting the rate of decomposition and the production of CO₂ (Roulet *et al.* 1991; Moore *et al.* 2011; Bhullar *et al.* 2013; Armstrong *et al.* 2015). While several anaerobic respiration pathways produce CO₂, decomposition rate, and hence CO₂ production, is higher in the oxic layer above the water table. Methane production, on the other hand, occurs exclusively by methanogens in the anoxic layer primarily found below the water table (Moore and Dalva 1997; Dinsmore *et al.* 2009; Bhullar *et al.* 2013; Yavitt *et al.* 2019). Due to the relationship between CH₄ emissions and anoxic conditions, seasonal fluctuations in the water table and soil and air temperatures tend to be reflected in CH₄ emissions, which are often small in the spring, growing to peak emissions in the summer, and decreasing into the fall (Moore *et al.* 2011). While water table is an important predictor of seasonal CH₄ fluxes, other variables such as soil temperature and vegetation can also be drivers (Bubier *et al.* 1993; Dunfield *et al.* 1993; van Hulzen *et al.* 1999; Treat *et al.* 2007). Interestingly, the relative importance of each variable depends on the time scale of measurements (Treat *et al.* 2007; Moore *et al.* 2011). Water table, for example, is often considered a driver of CH₄ fluxes, however on short time scales – hours or days – the relationship can be challenging to identify. When water table and CH₄ fluxes are observed over months or seasons, the

lag effect is muted, and patterns may be clarified. Since C fluxes are closely related to the water table, long-term behaviour of the water table has important implications for fluxes from peatlands to the atmosphere.

While CO₂ and CH₄ production occur in different areas within the peat column, both begin with the physical decomposition of dead organic matter (van Hulzen et al. 1999). This is typically first carried out by microfauna in the surface peat followed by leaching downward with water. This physical step breaks organic matter into more easily accessible molecules for microbes to utilize such as cellulose, hemicellulose, and lignin (Tolunay et al. 2024). Although this form of C is more accessible, it often cannot yet be used by microbes, so the more complex polysaccharides are broken down into monosaccharides via exocellular enzymes released by microbes into the environment (van Hulzen et al. 1999; Bridgham et al. 2013; St James et al. 2021; Tolunay et al. 2024). For instance, cellulose found in plant cell walls is a complex chain of molecules, so exocellular enzymes are released by microbes to break the hydrogen bonds between the glucose monomers within cellulose through hydrolysis (Bridgham et al. 2013). Once the glucose is accessible, decomposition proceeds either via ethanol fermentation or acetogenesis (St James et al. 2021). During ethanol fermentation, one mole of glucose is turned into 2 moles of ethanol, with 2 moles of CO₂ as a byproduct (Bridgham et al. 2013). During acetogenesis, one mole of glucose is converted into 2 moles of acetate, with 2 moles of CO₂ and 4 moles of H₂ as a byproduct (St James et al. 2021). At this point, decomposition depends on the position of the water table. Above the water table, the byproducts of fermentation and acetogenesis (CO₂ and H₂) are released into the atmosphere and surrounding environment (St James et al. 2021). Below the water table, methanogens, or microbes that produce CH₄ as a byproduct during microbial metabolism, and are strict anaerobes, are present. Methanogens capable of utilizing CO₂ and the other byproducts

of fermentation (i.e., acetate, ethanol, etc.) will then consume it, and several complex processes during either ethanol fermentation or acetogenesis will proceed (St James et al. 2021). Interestingly, a compound released from Sphagnum mosses called sphagnum is thought to prevent methanogenesis, sulfate reduction, and CO₂ production (Stalheim et al. 2009). This may increase the presence of acetate within the peat column, lowering the pH of the system, which is advantageous to Sphagnum mosses, as decomposition is further slowed by the acidic environment inhibiting the growth of acetogenic methanogens (Stalheim et al. 2009). Under acetoclastic methanogenesis, acetate is converted into CH₄ and CO₂ (St James et al. 2021). During hydrogenotrophic methanogenesis, 1 mole of CO₂ and 4 moles of H₂ are converted into 1 mole of CH₄ and 2 moles of H₂O (St James et al. 2021). There is also growing evidence that methylotrophic methanogenesis that uses, for example, methanol or methamine as substrate, is an important CH₄ production pathway in peatlands (Kurth et al. 2019). Once produced, CH₄ can be oxidized to CO₂ by methanotrophs in the presence of oxygen or other electron acceptors (Tolunay et al. 2024).

After it is produced, CO₂ enters the atmosphere by diffusion, rising along a concentration gradient between peat and the atmosphere. CH₄ can enter the atmosphere one of three ways: ebullition, plant-mediated transport, and diffusion through the peat column (Bridgham et al. 2013). During ebullition, the CH₄ rises in bubbles that have formed below the water table once methanogenesis causes the partial pressure of dissolved gases in solution to be greater than the hydrostatic pressure of peat (Lai 2009). Gas bubbles rise due to the density gradient between the peat and the atmosphere, often coaxed from the peat by changes in pressure (Comas et al. 2011; Bridgham et al. 2013). In the case of plant-mediated transport, CH₄ is taken up by plant roots that reach below the water table and is moved through the plants by structures called aerenchyma, which passively transport CH₄ via diffusion due to

concentration gradients between the peat and the atmosphere (Bhullar et al. 2013). Yet plant-mediated transport does not exclusively facilitate transport upward, since the concentration gradient of O_2 encourages downward transport. As a result, O_2 can be taken into a plant from the atmosphere through the aerenchyma and passed into the zone below the water table by roots (Moore and Dalva 1997; Dinsmore et al. 2009). In this instance, the CH_4 oxidation potential is increased by plant mediated transport (Dinsmore et al. 2009). Methane diffuses along a concentration gradient from below the water table to the atmosphere, first passing through the portion of peat located above the water table (Clymo 1984; Bridgham et al. 2013). Due to the time spent above the water table, diffusing CH_4 is subject to oxidation, with estimates suggesting that 22% of diffusing CH_4 may be oxidized within a 10 cm span between the water table and atmosphere (Pearce and Clymo 2001). Oxidation remains a possibility below the water table as well, with some estimates suggesting 90% or more of the CH_4 produced below the water table may also be oxidized before it reaches the oxic zone (Bubier et al. 1993).

1.4 Assessing varied responses to burn severity

Sphagnum mosses are characteristic of northern peatlands and their partially decayed remains often comprise much of the peat column. Sphagnum retains a high moisture content (McCarter and Price 2014) and exerts a profound influence in the position of the water table and maintenance of near-surface moisture content within peatlands. Distinct Sphagnum species have differing water storage capacities, and as a result, hollows tend to burn more readily than hummocks as the Sphagnum species on hummocks often have a greater capacity to retain water (McCarter and Price 2014; Turetsky et al. 2015). Within peatlands, post-fire microtopography can vary depending on the severity of burn. Less severe fires are more likely to preserve hummocks, whereas more severe fires, which burn at higher temperatures, can lead to the leveling of microtopographic features (Nelson et al. 2021). This is due to

the energy transfer of heat into the water that creates more suitable fuel by drying the peat and allowing the fire to burn farther horizontally than a less severe fire. After fire, hummocks show greater pore water tension, and are likely to cause hydrologic imbalance between peatlands and their surrounding landscape (Thompson et al. 2014). When peatlands become hydrologically isolated, they become more susceptible to repeated burning.

Estimates of immediate C lost to combustion range from 600 g C m^{-2} – 2500 g C m^{-2} (Turetsky et al. 2011; Wilkinson et al. 2023) in pristine peatlands. Meanwhile, peatlands acquire an estimated $3.8 \text{ g C m}^{-2} \text{ day}^{-1}$ – $6.9 \text{ g C m}^{-2} \text{ day}^{-1}$ (Sulman et al. 2010). It is estimated that peatlands return to their C acquisitive function within 10-13 years (Wieder et al. 2009). In the boreal region of Canada, fire return intervals are estimated to be around 123 years (Wieder et al. 2009), however, if there is a reduction in the wildfire return interval to 61 years or less, peatlands are expected to act as net sources of C. As the climate becomes warmer and drier under current projections, deeper burning which leads to increased losses to combustion, or a shortened wildfire return interval, will become more prevalent. With severe burning, above- and below-ground biomass and peat are lost to the atmosphere as GHG fluxes (French et al. 2002). Fire severity alters surficial soil properties, increasing hydrophobicity and bulk density (Thompson and Waddington 2013); radiative flux, increasing exposure to direct sunlight (Brown et al. 2015; Li et al. 2017); hydrology, with lower canopy interception and evaporative losses from the soil (McCarter and Price 2014); and soil microbial communities, decreasing microbial load (Orwin and Ostle 2012; Turetsky et al. 2014). In these ways, fire severity alters peatland functioning, but few studies have assessed the influence of burn severity on the timing of recovery (Nelson et al. 2021). This is a necessary development as fires become more severe and understanding of peatland functioning during recovery is forced to change.

1.5 Post-fire dynamics: environmental responses and plant community adaptations

Fires affect the typical functioning of peatlands, making them more susceptible to repeated burning as they become hydrologically isolated, have increased bulk density, and may be prone to shrubification, especially in the margins of peatlands (Kettridge et al. 2015; Nelson et al. 2021). Plant successional changes are initiated with burning, which increases nutrient availability and temperature in the soil (Turetsky et al. 2014; Kettridge et al. 2015). Successional changes traditionally led peatlands to return to their pre-burn community composition over time; however, it is expected that with a deepening water table in response to climate change, vascular plants may take the place of *Sphagnum* species (Laine et al. 2021). Vascular plants are better adapted to tolerate temperature changes and do not rely exclusively on capillary action to access water, as mosses do (Laine et al. 2019). Due to alterations of the water table, this change is projected to decrease C accumulation, increase decomposition, and increase soil respiration.

Peatlands exhibit resilience mechanisms against severe burning, for example, hydrophobicity in feathermoss-dominated peatlands enhances peatland resilience to fires, by reducing evaporation following wildland fires (Kettridge et al. 2014). Feathermosses have been found to have a degree of hydrophobicity, which is thought to be capable of limiting water loss to surface evaporation by reducing near-surface soil moisture. This may prove strategic, especially in the context of the threat from more frequent and severe drought, which increases peatland vulnerability to wildfire. Hydrophobicity is not observed as prominently at *Sphagnum*-dominated sites, which may increase vulnerability to deep water table positions during times of drought, or following disturbance (Kettridge et al. 2014). For example, recent research (Wilkinson et al. 2018; Nelson et al. 2021) suggests that *Sphagnum fuscum* is more

resistant to combustion during wildfire but is less resilient after fire due to its delayed hydrological recovery. Water repellency has been found to be greater at the near-surface than directly at the surface (Kettridge et al. 2014), which suggests that peatland plants like feathermosses have mechanisms to limit evaporation. Burn sites have a more severe degree of water repellency than unburned sites, which is even more pronounced in feathermoss sites (Kettridge et al. 2014); however, with deep peat burning, this strategy may prove disadvantageous. A transition from *Sphagnum*-dominance to feathermoss-dominance increases the risk of repeated burning because of a lower moisture content following disturbance, as the hydrophobic behaviour of feathermosses may prevent precipitation from infiltrating (Wilkinson *et al.*, 2018a, Nelson *et al.*, 2021). Thus, feathermoss-dominated sites may be better at preventing wildfire but are more susceptible to repeated burning due to the same mechanisms.

Interestingly, Elmes et al. (2019) found the opposite relationship to be true in a study of a fen watershed in Northern Alberta, where water repellency in burned organic soils was significantly lower than the unburned control. In this study, Elmes et al. (2019) argue that soil hydrological properties may not be applicable to every instance, and control by another variable is likely at play, such as vegetation, air or soil temperature, or fire temperature and duration. As well as decreased hydrophobicity, the burned site had significant effects revealing higher infiltration rates, higher mean bulk density, and lower water retention. As bulk density and infiltration rates increase, a lower volume of water has a higher impact on the water table. As fires burn through peatlands burning the top layers of peat, the soil characteristics shift from having low bulk density, high infiltration rates, and low water retention abilities to soil with high bulk density, low infiltration rates, and high water retention abilities. It is because of these characteristics that peatlands can withstand decreases in water table – they are capable of transporting soil moisture toward the surface via capillary action. The result of changes to the soil

and hydrological properties of peat is flashier water tables, which, when combined with the hydrophobicity Elmes et al. (2019) noted, may increase the risk of drought in burned peatlands.

Peat fires remove vegetation; altering the energy balance by exposing soil to incoming radiation, altering long wave radiation; and increasing exposure to wind, changing latent heat fluxes (Li et al. 2017). The removal of vegetation influences soil moisture and specific heat capacity, as well as soil biogeochemistry by altering decomposition and nutrient uptake and causing changes in evapotranspiration regimes (Brown et al. 2015). Removal of vegetation has different impacts depending on peatland type. Fens tend to be more responsive to changes in water table than bogs, since bogs are precipitation-fed, often undergoing periods of drought; however, fens are more susceptible to invasive species due to their greater nutrient availability (Eskelinen and Harrison 2015; Laine et al. 2019). With alteration to soil moisture, fens may be considered more resilient, however, by exposing soil and creating space for new, possibly invasive species to propagate, bogs may be considered more resilient. It is challenging to predict the effect of fire on peatlands, as there are numerous intricacies that may alter the recovery trajectory.

Not only do plant community compositions change, but there are physiological changes to plants themselves that may be compared between burned and unburned areas. These measurements are called plant functional traits and can be used to make inferences about how a plant utilizes resources, and whether it uses resources differently in different environments. It has been found that changes to the C cycle have led to physiological changes in functional traits (Li et al. 2021). As such, these traits can only be measured in cases where the plant community composition has not diverged too greatly from its pre-burn composition. Since wildfire often results in increased light and nutrient availability, it is expected that plant functional traits will differ from burned areas to unburned areas. Although there

is no research specifically on the effect that burning in peatlands has on plant functional traits, there are studies that either observe wildfire in different ecosystems, or other forms of disturbance in peatlands. Thus far, many studies suggest that specific leaf area, a plant functional trait which is the ratio of total leaf area to its dry mass, increases in response to disturbance, while leaf dry matter content, the ratio of fresh mass to dry mass, decreases (Wilson et al. 1999; Kotowski et al. 2013; Smart et al. 2017; Konings et al. 2019). These findings suggest that plants may change their resource allocation strategies in response to disturbance to a rapid growth strategy, rather than a C accumulation strategy which is observed in comparable undisturbed ecosystems (Kotowski et al. 2013; Konings et al. 2019). Not only are physical functional trait differences detected across sites, but chemical traits as well. Li et al. (2024) observed that the chemical makeup of plants was altered by burning, and by measuring C:N was able to determine the completeness of combustion, with the potential to identify the type of combustion that had occurred.

Some plants are more closely related to the C exchange within peatlands than others. For example, Moore et al. (2011) noted that collars which had a significant proportion of sedges, particularly *Eriophorum vaginatum*, had a 3–4-fold increase in CH₄ emissions compared to collars without sedges. Since plants with aerenchyma are capable of transporting CH₄ passively from their roots to the atmosphere, in times of high water table, more CH₄ is being produced, which can be transmitted from the soil through the aerenchymatous tissue of the plants. A similar relationship to that of CH₄ efflux and *E. vaginatum* was also noted with *Maianthemum trifolium* (Moore et al. 2011). Another example lies in *Picea mariana*, as with burning it is expected that some peatlands will shift to upland trees such as those in the *Populus* and *Betula* genus's, which store less aboveground C than conifers that currently dominate boreal peatlands (Brown et al. 2015; Kettridge et al. 2015).

Despite studies that have explained the role of vegetation on peatland fire, post-fire C emissions, and successional patterns, there is a gap in the knowledge of quantifiable traits that might provide insight into the rate of functional recovery in peatlands disturbed by fire, where recovery is the point in which the burned peatlands return to the same functioning as unburned peatlands. Analyses of plant functional changes related to fire regimes is crucial in understanding wildfire return intervals and may inform earth system models of C cycling.

1.6 Changes in carbon cycling to post-fire conditions

While the importance of understanding C fluxes and ecosystem changes during fire cannot be overstated, this project focuses on recovery, specifically six to seven years after a fire, with no repeated burning. Recovery following wildfire will be driven by the fires' characteristics, including burn severity, or depth of burn.

While peat burning has been previously studied, there is still much that is unknown. There is uncertainty regarding total C released during peat burning and following fire cessation due to several variables. First, C emissions vary geographically, and can be significantly different depending on location. For example, studies are often conducted in North America, where peat burning is common during high intensity wildfires (Rogers et al. 2015), which may lead to an overestimate of total C emissions when applied to fires in Eurasia (Rogers et al. 2015), that typically only burn above ground biomass. Although latent heat transfer is high from Eurasian wildfires, the soil does not often burn (Thompson et al. 2014). Second, depth of burn typically increases throughout the fire season, but this is only the case with small upland forest fires (Turetsky et al. 2011). With large fires in peat and permafrost regions, deep burning often occurs throughout the fire season (Turetsky et al. 2011).

Another source of uncertainty is the role of pyrogenic C in total emissions estimates. An estimated 50% of pyrogenic C produced by aboveground biomass continues to hold C for centuries, although it is underrepresented in the global C cycle (Santín et al. 2016). High-temperature fires create pyrogenic C that has increased resistance against degradation compared to the same organic matter pre-burned, while low-temperature fires create more water-soluble and biodegradable pyrogenic C (Santín et al. 2016). On average, however, pyrogenic C has a longer residence time in the environment than unburnt vegetation and can be considered a C sink on the decadal or centennial time scale (Santín et al. 2016). The difficulty in assessing the total C stored in pyrogenic C post-fire is that sites must have comprehensive fuel data prior to burn to understand the amount of fuel converted to pyrogenic C (Santín et al. 2016), which has not yet been accomplished. Not only is the presence of pyrogenic C difficult to quantify, but many forms are challenging to detect, such as belowground/root burning that leads to the creation of pyrogenic C, or ash that has formed and been deposited elsewhere due to wind (Santín et al. 2016; Kane et al. 2019), often before field studies of burned sites are even able to begin.

Little is known about rates of C emissions from pyrogenic C in the time following fire, nor how it is released. This concept has direct implications for the rate of recovery of peatlands, however it is unclear if appreciable quantities of pyrogenic C is stored in peat nor how long it lasts. Furthermore, the influence that pyrogenic C has on fluxes relative to unburned peat has not been characterized. It is crucial to determine the impact of pyrogenic C on the recovery period of burned peatlands as it will inform recovery rates, especially with repeated burning becoming more common.

Traditionally, it has been thought that following a fire, the C lost by burning is regained by the growth of vegetation within a few years (Turetsky et al. 2015; Van Der Werf et al. 2017). With boreal forests trending toward being hotter and drier than they have ever been before, fire is burning into

deeper peat stocks, accessing and releasing C to the atmosphere that might have been held for millennia within peat deposits (Roulet et al. 1991; French et al. 2002; Turetsky et al. 2004; Turetsky et al. 2011; 201; Hu et al. 2019). Even if rapid vegetation growth immediately after fire offsets increased emissions, decomposition and ecosystem respiration are likely to increase long-term C losses (Nelson et al. 2021). Wildfire causes an immediately shallower water table because of a reduction in the peat surface elevation; however, the long-term effects of fire under climate change conditions may lead to deeper water tables (Nelson et al. 2021). A deeper water table results in greater decomposition as microbes, fungi, and invertebrates that require oxygen to survive can access and release the C held within soil organic matter, increasing ecosystem respiration.

1.7 Impacts of climate change - altered dynamics and increasing fragility

With changes to the world's historic climate patterns, fires in the boreal region are projected to become more common, and more severe (Turetsky et al. 2015). More severe burning leads to the release of GHGs from organic matter in deep peat that has been stored for centuries to millennia (Turetsky et al. 2014). Fire affects the climate and energy budget in two main ways: (1) emissions of trace gases and aerosols and (2) alterations to terrestrial ecosystem states and functioning (Bowman et al. 2009). Wildfire is the primary form of terrestrial ecosystem disturbance globally (Bowman et al. 2009). As such the changes made by fire on an ecosystem need to be thoroughly understood so that projections of future fire regimes provide accurate estimates of land-based GHG emissions and C storage.

Many studies have observed the influence of warming and drying on peatlands. Some argue that increased afforestation due to warming may increase biomass losses to fire (Wilkinson et al. 2018; Nelson et al. 2021), that more nutrients will become mobile in the systems because of decomposition

(Laine et al. 2018), or that ecosystem respiration will increase (Laine et al. 2018). Across all studies, however, warming and drying in peatlands leads to a greater release of GHGs, likely through multiple mechanisms (Li et al. 2021). The effects of climate change may differ depending on wetland type. For example, during a dry season, a bog will receive minimal amounts of water through precipitation (Zoltai and Vitt 1995). A fen, on the other hand, will remain connected to the groundwater supply (Zoltai and Vitt 1995). This difference in water supply is often what increases peatland vulnerability to burning (Laine et al. 2019), which seems to suggest that bogs may be more vulnerable, as well as subject to deeper burning, than fens. Although fire does not selectively or preferentially burn peatlands, no matter the classification (Turetsky et al. 2004), it is possible that ecosystem mechanisms that discourage burning could be the difference in whether a fire will ignite or not (Turetsky et al. 2004). In this way, one is led to assume that with climate change, bogs will become increasingly vulnerable, and fens may be more resilient.

Throughout the literature, it is noted that CO₂ exchange is more closely related to the effect of drying than the effect of warming (Laine et al. 2018; Swails et al. 2022). The impact of drying is even further accentuated in previously burned peatlands, whose water tables are subject to higher variability than unburned peatlands (Nelson et al. 2021). On average, 23,000 ha yr⁻¹ burn in Albertan peatlands (Turetsky et al. 2004). These peatlands are not only then releasing vast deposits of terrestrial C to the atmosphere but are also more likely to burn repeatedly and burn more deeply into the peat deposit. Seasonality is a large contributor to changes in moisture regimes, as seasonal ice thaws and the depth of ground layer burnt increases due to greater fuel availability (Turetsky et al. 2011). As fires can burn deeper due to changes in moisture, deep plant roots and fungal communities may also be subject to burning (Turetsky et al. 2004), leading to lower ecosystem diversity, and an increase in more readily

combustible fuels. Such events may also lead to a loss of hydrologic regulation in the system, which not only increases the risk of repeated burning, but also reduces the rate of peat accumulation, and possibly increases the rate of decomposition in the peat column (Turetsky et al. 2014).

The water table acts as a proxy for the environmental conditions that dictate CH₄ production and consumption. CH₄ oxidation, a result of water table decline, is increasing in peatlands, as CH₄ production is closely related to the moisture content within the peat column (Roulet et al. 1991). Although it is noted throughout the literature that changes in peatland CH₄ cycling are more closely related to drying, warming because of climate change also influences C exchange (Davidson et al. 2019). Yvon-Durocher et al. (2014) observed an exponential relationship between CH₄ oxidation and soil temperature, which is closely related to air temperature, especially in near-surface peat, a relationship that is strengthened in burned peatlands (Brown et al. 2015). Rapid and vast changes in burned peatland's soil temperatures are a result of increased exposure, which is enhanced by a lack of protective litter cover, causing increased insolation which previously would have prevented rapid changes in daily soil temperatures (Brown et al. 2015).

1.8 Remaining questions and knowledge gaps in peatland wildfire research

Although research focused on boreal fire regimes has been taking place for many years, the net effect of fire on the C cycle is still unknown (Bowman et al. 2009; Hantson et al. 2016; Santín et al. 2016; Van Der Werf et al. 2017; Lasslop et al. 2019). As a result, C cycling models and climate models are poorly informed, and may over or underestimate the true global C flux (Ballhorn et al. 2009; Nelson et al. 2021). There are several processes that remain challenging to include in modelling currently, including longitudinal complexities, peat smouldering, small fires, El Nino events, depth of peat, depth

of burn, influence of pyrogenic C, and differentiation amongst peatland classification (Hashimoto et al. 2004; Eskelinen and Harrison 2015; Rogers et al. 2015; Santín et al. 2016; Laine et al. 2019; Nelson et al. 2021). It has been suggested that boreal ecosystem C fluxes may emit 3-10 times more C than the values currently used in general circulation models (Harden et al. 2000).

Geographical differences in fire regimes have been studied in terms of latitudinal variation, for example, representing boreal and tropical peatland fires independently. However, longitudinal variation is often overlooked, with the unsubstantiated assumption that all peatland fires of the same latitude produce similar fluxes of C. Boreal fires behave very differently in North America and Eurasia, despite similar climatic conditions (Rogers et al. 2015). North American forests tend to support high-intensity crown fires due to presence of vegetation evolved to be consumed by fire as part of their life cycle (Rogers et al. 2015). Trees such as *Pinus banksiana*, which have serotinous cones evolved to release seeds under increased temperatures are an example of such evolutionary traits. This is not the case in Eurasia, where species have evolved to resist burning by self-pruning lower branches, keeping fires principally in the canopy (Rogers et al. 2015). It is important to consider species level traits in global evaluations of the effects of wildfire on GHG emissions, as fire severity and fuel source may significantly impact emissions and recovery and have different implications for climate change modelling.

Peat smouldering has long been a challenge in global fire models, as it is difficult to track both with satellite imagery, and from the ground. Peat smouldering produces far greater emissions than flaming combustion, but due to the physical complexities of identifying peat smouldering, it is underrepresented and poorly understood (French et al. 2002; Van Der Werf et al. 2017; Hu et al. 2019). Not only are areas vulnerable to and undergoing peat smouldering difficult to track, active peat

smouldering has even greater challenges associated with tracking (Van Der Werf et al. 2017). Often, peat fires remain undetected until resurfacing as flaming combustion, at which point they have burned through ancient C stocks and released vast amounts of C to the atmosphere (French et al. 2002).

Small fires are difficult to identify and track, especially considering the impact of peat smouldering, and the capacity for fire to ignite and extinguish kilometers from the original ignition site seemingly spontaneously (Van Der Werf et al. 2017). New techniques are being suggested, including Moderate Resolution Imaging Spectroradiometer (MODIS), which is capable of accounting for burned area, thermal anomalies, and surface reflectance, which combined, provide an estimate of total burned area (Van Der Werf et al. 2017). MODIS is one of the best pieces of equipment when attempting to quantify total burned area, increasing estimates by as much as 26% (Van Der Werf et al. 2017). Although this leads to greater accuracy in terms of total burned area, it is important to keep in mind the potential underestimation brought on with underrepresented smouldering, spatial variation, and so on.

El Nino events and other abnormal climate events provide yet another challenge when modelling C cycling (Van Der Werf et al. 2017). Models are used to create accurate projections of future conditions; however, when conditions such as climate, often an underlying condition in modelling C cycling, are variable, it creates even more uncertainty. The difficulty is that these events do not look the same year after year, they cause seemingly random changes to air temperatures, vapor pressure deficits, short wave radiation, etc., meaning that they cannot be easily modelled as the interannual variability of El Nino events is different year after year (Hashimoto et al. 2004). Although these events provide difficulties for models, some interesting patterns have been observed. In the context of El Nino, Hashimoto et al. (2004) found that C sequestration tended to be higher with cooler

temperatures, low vapor pressure deficits, and high radiation, while C release increased under opposing conditions

Most models are also guilty of underestimating total depth of burn, and thus underestimating total emissions. Typically, fire is thought to burn the top 15 cm of soil; however, based on field samples and LiDAR data, depth of burn is often much deeper (Nelson et al. 2021). Depth of burn may be misrepresented because of a lack of understanding of peatland soil variability. For example, there is likely to be great variability between the shallower peat margins, and the thick middle peat deposits of peatlands (Nelson et al. 2021; Mayner et al. 2024).

Another source of uncertainty is the effect of coarse woody debris decay following fires (Amiro et al. 2010). When fires first burn an ecosystem there is an immediate efflux of C from burning biomass and soil. Many decomposable materials remain upright for years, avoiding the moist, warm soil surface that is conducive to decay (Amiro et al. 2010). When this biomass does eventually fall over, the efflux from decomposition is unknown. This is also related to decomposition rates of pyrogenic C, which are not well understood, and whose impact on the recovery rate in burned peatlands is also unknown (Armstrong et al. 2015; Santín et al. 2016).

As a final challenge to fire modelling, pyrogenic C is not yet considered in most C budgets and global warming assessments (Santín et al. 2016). The impact of pyrogenic C, in contrast to other variables, may overrepresent the C released as fuel during wildfire. Carbon often remains in its pyrogenic form for decades to centuries. Consequently, models that do not account for this long-term storage may overestimate C emissions.

Creation of a more accurate fire model that includes new estimates of total burned area including updated depth of burn, small fires, peat smouldering, and modifications to fuel consumption based on spatial variation that represent latitudinal and longitudinal variations is necessary for accurate estimates of future climatic conditions and current C cycles (Rogers et al. 2015; Van Der Werf et al. 2017). In addition to these uncertainty, tracking the impact of fire on ecosystem C storage over decade to century time scale requires an understanding of the response of that ecosystem to the fire disturbance and recovery post-fire. Currently, following a fire the response of peatland C fluxes and recovery trajectory are unclear but are necessary to model long-term C cycling of these ecosystems, particularly in response to changing fire regimes. Current estimates place the recovery period at 10-13 years (Stocks et al. 2002; Wieder et al. 2009). Few studies observe short term recovery, and often do not consider the impact of fire severity on fluxes and recovery. Many studies also tend to address only one peatland type, typically bogs, which is applied with a broad stroke to all other peatland types. The difference between bogs and fens in terms of wildfire recovery is not well known, and despite fens accounting for most peatlands across the boreal regions of Canada, they are often neglected in the literature on wildfire recovery. Thus, the objectives of this study are to (1) assess differences in plant community composition and functional traits between two peatland types, a bog and a fen, along a fire severity gradient and (2) quantify and compare surface level post-fire C fluxes between two peatland types, a bog and a fen. Results from this study will help fill knowledge gaps on peatland C cycling following fire, including differences between bogs and fens, while also providing information on process-based controls on these post-fire C fluxes that can inform future peatland C cycling models that include wildfire.

Chapter 2

Vegetation composition and function in peatlands post-fire: plant community composition and plant functional traits

2.1 Introduction

Boreal peatlands are globally significant sinks of carbon (C), as the long-term rate of organic matter input is greater than the rate of decomposition and combustion (Turetsky et al. 2014). Carbon accumulation rates, as well as soil quality are dependent on a peatlands' vegetation community composition (Tolunay et al. 2024). For example, fen peat tends to have higher ratios of *Carex* species roots as organic matter inputs, while bog peat tends to contain a much higher proportion of *Sphagnum* mosses (Clemmensen et al. 2013). Non-vascular vegetation is especially important to peatland function, as mosses are often key in moderating the water table and generating peat soil properties (Kettridge et al. 2014; Lukenbach et al. 2016). Vegetation is a critical aspect of peatland functioning; however, few studies assess the response of vegetation to disturbance by contrasting pre- and post- disturbance vegetation community composition, and differences in resource allocation strategies and vegetation quality. Given this, it is crucial to understand the impact that wildfire, the most prevalent peatland disturbance (Brown et al. 2015), has on the vegetation community

Plant functional traits are measurable traits that elucidate how a plant reacts to a given environmental condition and how plants control ecosystem functioning (Laine et al. 2022). Plant functional traits can be used to inform resource allocation strategies, and thus can be used to indicate whether plants allocate resources differently based on environmental conditions and in response to disturbance. Common plant functional trait measurements tend to be either quantitative, including specific

leaf area (SLA; leaf area divided by dry mass), leaf dry matter content (LDMC; leaf fresh mass divided by leaf dry mass), and height, or can be qualitative, assessing plant life strategies such as whether a plant seeds or resprouts (Pausas and Keeley 2014), or whether a plant has storage organs such as tubers. Application of plant functional traits to peatland studies is relatively new, but recent research suggests that plant functional traits may be indicative of the C storage capacity of an ecosystem (Goud et al. 2017 Sep; Jassey et al. 2018 Mar; Laine et al. 2022), as well as a sign of recovery following disturbance (Kotowski et al. 2013).

Based on a critical review of the literature, no studies to date have observed plant functional traits in peatlands disturbed by fire. Studies of disturbance have been conducted largely on conserved open fens in Europe, where measures are often taken to prevent succession, maintaining open fen structures (Amiro et al. 2010), or in Mediterranean landscapes in Southern Australia, where fire is a common disturbance event (Enright et al. 2014). In boreal Canada, these practices are not necessary, as human intervention following disturbance is rare, typically leaving peatlands to be regenerated by natural processes, or following other practices such as mounding, as hydrological connectivity remains intact (Elmes et al. 2021; Kleinke et al. 2022). Furthermore, fire regimes differ markedly between landscapes (Rogers et al. 2015), so while inferences can be made based on the findings of the literature, they will remain speculative until field research can be conducted. The primary source of disturbance to Canadian wildlands is fire (Bowman et al. 2009) making it a crucial point of study to understand recovery and site behavior following disturbance, especially in landscapes where C storage is high and potential losses from fire may be catastrophic. Currently there is a need for more research into the effects of variation in burn severity, the recovery trajectory of different peatland classifications following fire, and the general ecological impacts of wildfires in peatland ecosystems.

When fires burn through peatlands, the outcome differs depending on the type and severity of the fire. Under a moderate fire regime, often only the overstory burns, and ground layer vegetation may remain intact. However, when a severe fire occurs, aboveground biomass burns and little to no living plant life is left behind (Davies et al. 2023: 202). Regeneration is reliant on the seed bank and associated microbial communities within the soil (Ward et al. 2012). Post-fire, access to light, labile nutrients, and water is increased, and surficial soil properties are altered so that evaporative losses are lowered, bulk density is increased, and immediately following fire the water table is much closer to the peat surface (Thompson and Waddington 2013; Kettridge et al. 2014). Fires do not always burn equally across a peatland. Microforms called hummocks and hollows affect the way fire burns within a site. Hummocks, which are raised, mound-like structures within peatlands are typically drier because of their elevation, resulting in communities of peat-forming plants that prefer to grow in drier conditions. Hollows, which are depressions in the landscape are typically wet or waterlogged areas that allow for peat formation by providing anoxic conditions that prevent rapid decomposition of dead organic matter. Hummocks often consist of species that have a greater capacity to retain water (Thompson and Waddington 2013). During fire, when peatlands are typically dry and vulnerable, this trait can be beneficial, as passing fire requires a higher energy input to burn through the wetter hummocks, with the hollows around them tending to having a greater depth of burn (Benscoter and Wieder 2003). In low to moderate severity fires, hummocks are retained within the peatland, and hummock-forming species can quickly regenerate (Benscoter and Wieder 2003). As fire severity increases, microform selectivity is reduced, burning deeper into the peat, both over hummocks and hollows, which may be because of pre-fire conditions being drier, or the effect of a hotter fire. No matter the pre-existing conditions, during severe fires, all microforms are burned and post-fire topography is less variable (Benscoter and Wieder 2003).

Although there have been no studies directly on peatland plant functional trait response to fire, there have been studies on other forms of disturbance that may inform how plants will respond to fire. Often studies are focused on one peatland class, generally bogs or fens, and rarely include both, as vegetation composition is known to be vastly different between the two ecosystems. However, data can be assimilated to determine overall trends. When peatlands undergo water level drawdown, bogs tend to use a resource conservation strategy. For example, an undisturbed bog, and a bog experiencing water level drawdown have been observed with LDMCs that differ, with undisturbed having greater LDMC than disturbed bogs (Laine et al. 2021). LDMC, which can be more easily understood as biomass by weight, is a practical measurement of plants' resource allocation strategies, with a greater value indicating a heartier plant, and a lower value indicating a life strategy of rapid expansion. On the other hand, fen plants shift to a stronger resource acquisitive strategy, for example LDMC measured at a disturbed fen by Laine et al. (2021) was greater than LDMC at an undisturbed fen. Bogs tend to be more resilient to large variability in the water table, having developed to tolerate the vagaries of precipitation events, whereas fens are less accustomed to drought due to a more consistent supply of ground or surface water. Therefore, fens may require vegetation communities to shift to adapt to changing conditions. Across both fens and bogs, plants tend to grow taller in response to drought, with a greater photosynthetic capacity compared to undisturbed ecosystems (Laine et al. 2021).

Other forms of disturbance may inform how peatland plant functional traits respond to fire. For example, Kotowski et al. (2013) found that active mowing in fen ecosystems led to greater SLA, which is measured as the ratio of leaf area to leaf dry mass. It is often used as an indicator of plant growth strategy, with a higher SLA value indicating more biomass allocated to total leaf area compared to lower values measured from the same species. Increased competition for light was responsible for the

higher SLA across all plants measured following mowing (Kotowski et al. 2013). This was true across all plants measured, as competition for light increased. SLA and LDMC were found by Konings et al. (2019) to be significantly lower and higher, respectively, in an open blanket bog compared to a forest-to-bog restoration site. Based on these studies, it appears that disturbances lead to increased SLA, and decreased LDMC, and similar patterns may be expected after a fire.

The objectives of this study are to quantify peatland recovery from fire using (1) vegetation community composition, inclusive of information on species presence-absence and abundance, and (2) plant functional traits of dominant species along a fire severity gradient. There are two main hypotheses: (1) seven years following a wildfire, moderately burned sites would be further along the recovery trajectory than the severely burned sites, meaning they would be more similar to the unburned treatments in terms of plant communities and plant functional traits, and (2) that the bog would be further along the recovery trajectory than the fen in terms of plant communities and plant functional traits.

2.2 Methods

2.2.1 Site location and description

The study was conducted in a moderate-rich treed fen (hereafter referred to as Poplar Fen; 56° 56' N, 111° 32' W; Figure 2) and an ombrotrophic bog (hereafter referred to as Prairie Creek Bog; 56° 57' N, 111° 30' W; Figure 2) (Figure 1). Prairie Creek Bog and Poplar Fen are located approximately 31 km south and 29 km north, respectively, of Fort McMurray, Alberta. Both sites were affected by the 2016 Horse River wildfire and have varying burn severities representative of the burn across each site. Portions of each study site were identified and instrumented with three burn severities, severely burned,

moderately burned, and unburned. Burn severities were based on relative depth of burn, where the deepest observed burns were referred to as severe burns, and respectively shallower burns were referred to as moderate burns. Burn depths ranged from 1 cm to 17 cm into the soil. In 2017, all three Poplar Fen sites were established and in 2022, were additionally instrumented for data collection. Prairie Creek Bog – Unburned, and Prairie Creek Bog – Moderately Burned were established and instrumented in 2022. In 2023, Prairie Creek Bog – Severely Burned was established. Since the site was established a year after the vegetation surveys were conducted, Prairie Creek Bog – Severely Burned was not included in the full vegetation surveys, or plant functional trait analyses, but is included in biomass measurements, collar vegetation surveys, and tree surveys. From May to August, the mean temperature in 2022 was 15 °C, and the total precipitation during the same period was 58 mm. During the same months in 2023, the mean temperature was 17 °C, and the mean total precipitation was 73 mm. Over the past decade, the mean temperature from May to August was 15 °C, and the mean total precipitation was 66 mm (Extracted from the Environment and Climate Change Canada Historical Climate Data web site (https://climate.weather.gc.ca/index_e.html) on December 18, 2023).

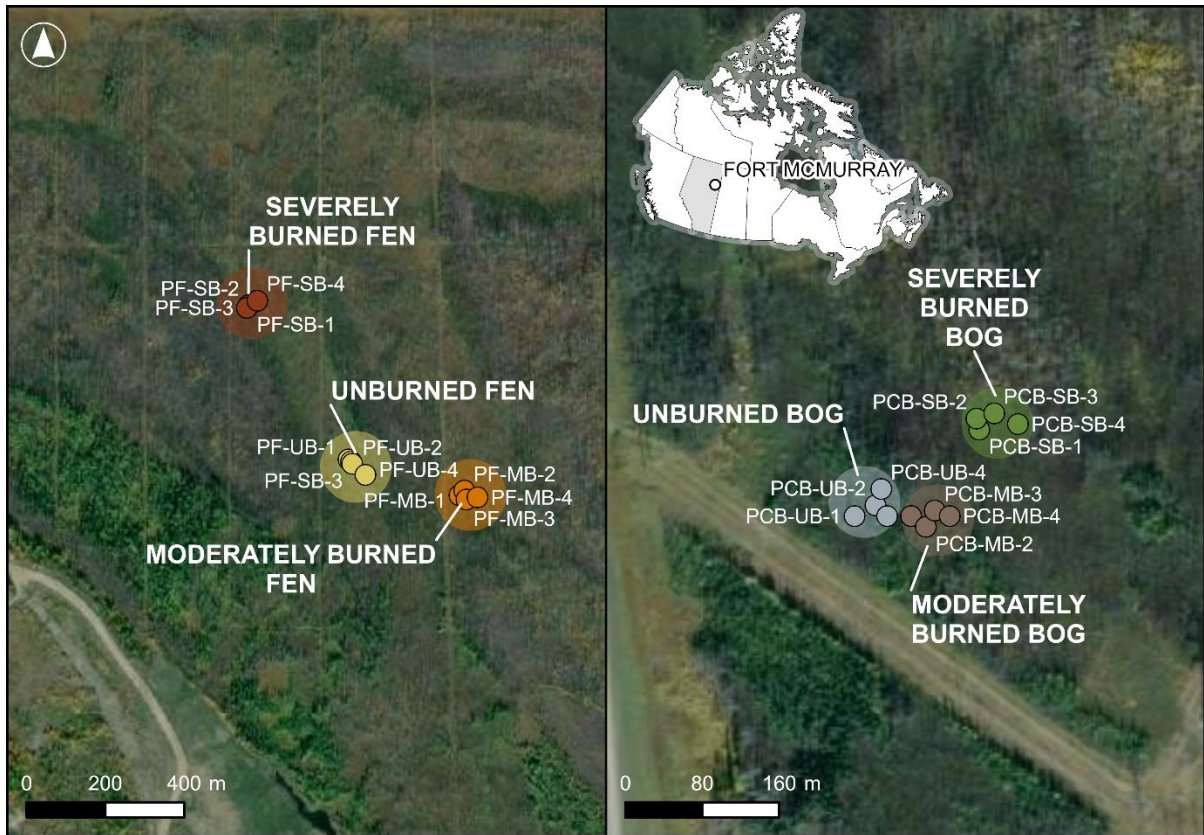


Figure 1: Map of Poplar Fen (left panel) and Prairie Creek Bog (right panel), which are approximately 60 km apart. Shown on the map are the sites along the burn severity gradient and collar locations for fluxes. Note the difference in scale between the two maps.

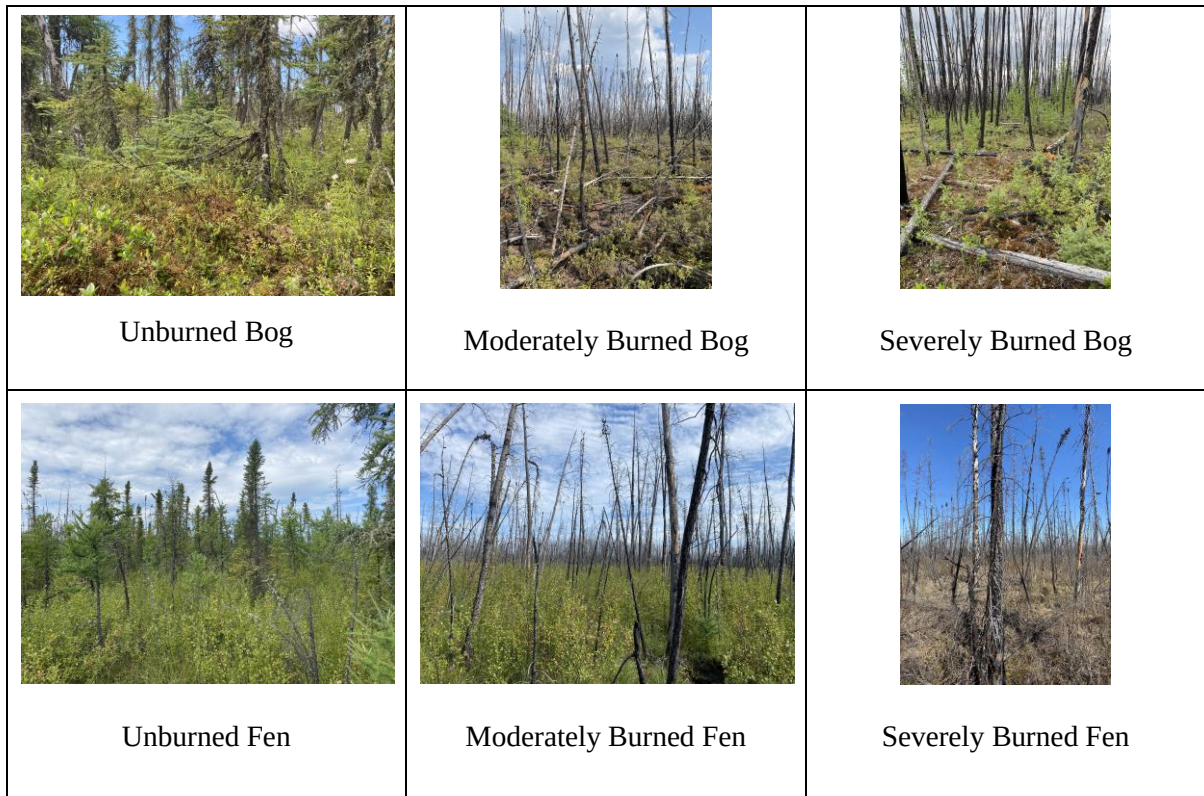


Figure 2: Photographs of each site within Prairie Creek Bog (top) and Poplar Fen (bottom)

2.2.2 Vegetation surveys

Round plots, hereafter referred to as collars, of approximately 20 cm in diameter were installed in hummock-hollow pairs. Four pairs were installed at each fen burn severity, and three pairs each were installed in the unburned and moderately burned bog sites. The collars were primarily used for C flux measurements (Chapter 3) but were also surveyed for vegetation cover. Vegetation within collars was identified to the species level using Johnson et al. (2017) and quantified visually into percent cover, so that the total vegetation in each collar may be greater than 100%.

Broader vegetation surveys across the study sites were conducted on August 2, 2022, August 3, 2022, and August 5, 2022. At each pair of collars, transect tape was run 24 m north from hummock

1A and 1 by 1 m quadrats were placed at 4 m, 9 m, 14 m, 19 m, and 24 m. Vegetation was measured by percent cover, where all species were identified using Johnson et al. (2017), which were then broken down into percentages of the total quadrat so that the species combined may make up more than 100%. After the vascular vegetation was measured, a 25 by 25 cm quadrat was placed in the bottom right-hand corner of the 1 by 1 m quadrat, where the bottom of the greater quadrat is along the edge of the transect tape. Ground cover of bryophytes and lichen was similarly measured, but in such a way that the total did not exceed 100%. Abiotic ground cover was also included, such as bare organic soil, rocks, and water. Plant functional group cover was calculated by summing species with each group including trees, shrubs, forbs, graminoids, and bryophytes, among others.

2.2.3 Biomass

Samples of biomass were collected near existing collars, on areas with comparable vegetation on the same microtopographic location. Samples were collected using a 20 cm diameter round plot (same dimensions as the collar), cutting out all existing vegetation, and placing the sample in a sealed and labelled paper bag to prevent rotting and loss of biomass while in transit. Vascular plants were cut at the moss surface, then the top 2 cm of moss were collected and assumed to constitute the biomass. It is difficult to determine the separation of living and dead moss biomass, but the top 2 cm generally includes all living material (Moore et al. 2002). Samples were separated based on species within 48 hours of collection, dried at 60 °C for 24 hours, and weighed in grams.

Shrub biomass was notable at the unburned and moderately burned fen sites and was not captured by the collar-sized biomass survey in a way that was representative of the site. Thus, an additional sampling strategy was used. Triplicate 50 by 50 cm quadrats were randomly placed at all

three fen sites into the 225 m² disposition described below. All shrub or tree seedling species that fell within the quadrat were cut at the base of their stem and put inside of a sealed and labelled paper bag to prevent rotting and vegetation loss during transit. Species were separated in-situ and later dried at 60 °C for 24 hours and weighed in grams.

Tree inventory surveys were conducted to determine the overstory biomass. Each measurement began at collar 1A. A tape measure was run 15 m north, then east, then south, then west, forming a 225 m² square. Trees were measured using a plastic caliper to determine either diameter at breast height (1.3 m above ground level) or basal diameter (for trees with a height less than 1.3 m). Measurements were recorded alongside the species identity. Tree measurements were then converted to biomass in kilograms using the allometric equations in Table 1 according to Martin et al. (2005) for *P. mariana*, *L. laricina*, *P. banksiana*, *P. balsmifera*, and *P. tremuloides*, and Huber et al. (2017) for all species where basal diameter measurements were recorded. Once the tree survey measurements were converted into grams, they were divided by the size of the plot they were measured in (225 m²) to obtain g/m². The sum of all tree biomass was used to represent the total tree biomass per site. The biomass values, tree biomass, vascular biomass, and understory biomass were combined to determine total biomass per site.

Table 1: Allometric equations for conversion of diameter at breast height (DBH, 1.3) to grams (g)

Tree species	Allometric Equation	Reference
<i>Picea mariana</i>	$= \frac{0.153 \cdot DBH^{2.248}}{m^2}$	Martin <i>et al.</i> , 2005
<i>Larix laricina</i>	$= \frac{0.1359 \cdot DBH^{2.298}}{m^2}$	Martin <i>et al.</i> , 2005
<i>Pinus banksiana</i>	$= \frac{10^{(-1.27 + (2.089(\log_{10} \cdot DBH))}}{m^2}$	Martin <i>et al.</i> , 2005
<i>Populus balsmifera</i>	$= \frac{10^{(-1.625 + (2.421(\log_{10} \cdot DBH))}}{m^2}$	Martin <i>et al.</i> , 2005
<i>Populus tremuloides</i>	$= \frac{10^{(-1.124 + (1.7(\log_{10} \cdot DBH))}}{m^2}$	Martin <i>et al.</i> , 2005
All basal diameter measurements	$= \frac{(0.035 + 0.206) \cdot (basal\ diameter^{2.577})}{m^2}$	Huber <i>et al.</i> , 2017

2.2.4 Plant functional traits

We chose to measure height, LDMC and SLA as these are commonly measured traits that indicate shifts in competition and resource acquisition (Wilson *et al.* 1999; Pekin *et al.* 2011; Goud *et al.* 2017 Sep; Smart *et al.* 2017; Konings *et al.* 2019; Daou *et al.* 2021; Laine *et al.* 2022), which were expected to change following wildfire. Functional traits were measured on three dominant species at each site; *Rhododendron groenlandicum*, *Picea mariana* and *Equisetum* spp. (*E. fluviatile* at Poplar Fen, *E. arvense* and *E. sylvaticum* at Prairie Creek Bog), as well as *Betula glandulosa* at Poplar Fen, but not Prairie Creek Bog, since it is dominant at the fen, but not present at the bog, and *Vaccinium vitis-idaea* at the bog, but not the fen since it is dominant at the bog but not present at the fen. Each plant's outstretched height at the tallest point was measured in-situ. Subsequently, 10 healthy leaves were collected from each specimen. Upon sample processing, plants were weighed and placed on a sheet of

paper with a measurement scale. Each plant was photographed in this fashion (Figure 3), then dried at 60 °C for ~24 hours, after which the dry weights were taken.



Figure 3: Examples of the photograph process for measuring plant functional traits. Photos include *Equisetum fluviatile*, *Rhododendron groenlandicum*, *Picea mariana*, and *Betula glandulosa*.

Using the photographs of the plants, ImageJ (Schneider et al. 2012) was used to calculate total leaf area per sample. For most plants, 10 healthy leaves were collected from the plant and photographed, however, since the physiology of *Equisetum* is contested, and the leaves are not necessarily practical to measure (Cody and Wagner 1980), the entire plant was analyzed for leaf area. These measurements were used to calculate

$$SLA = \frac{\text{average leaf area}}{\text{average leaf dry mass}}; \text{ and}$$

$$LDMC = \frac{\text{average leaf dry mass}}{\text{average leaf fresh mass}}$$

This data was compared across treatments to determine if functional traits differ between wetland classification or burn severities.

2.2.5 Data analysis

All data were analyzed using R version 4.2.2 (R Core Team (2022). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL <https://www.R-project.org/>). To assess the impact of wildfire on biomass, collar vegetation cover, and total site vegetation cover, linear regression models (LMs) were created with the fixed effect set as site. To analyse height, SLA, and LDMC, data was separated by species and LMs were run with the fixed effect of site to determine if functional traits differed for each species based on treatment (i.e., a combination of peatland classification and burn treatment). LMs were also created for percent cover by separating each functional group, and running an ANOVA to determine whether there was a significant effect of site. The LMs were created using the function ‘lm’ in base R and tested using the anova output with Type III sum of squares. If statistical significance with a confidence interval of 95% was given to site, the ‘emmeans’ function in the *emmeans* package (Lenth, 2023) was used with a Tukey adjustment for pairwise comparisons among treatments.

To calculate both the Simpson and Shannon diversity indices, the function *diversity* in the *vegan* (Oksanen et al. 2022) package was used. Simpson’s diversity index uses the equation

$$D = \frac{\sum n_i (n_i - 1)}{N(N - 1)}$$

where n_i is the number of individuals in the i -th species, and N is the total number of individuals in the community. Shannon’s diversity index uses the equation

$$H = -\sum P_i (\ln P_i)$$

where P_i is the proportion of each species in the sample. Numeric vegetation data was used. Statistical significance was not measured, as the sample size was too low.

2.3 Results

2.3.1 Vegetation community composition

The makeup of each site differed based on percent cover (Figure 4), but there were a few apparent trends. The moss percent cover was highest at the unburned sites, decreasing first to the moderately burned and finally to the severely burned. At Prairie Creek Bog, shrubs decreased in the same way as mosses, while bare organic soil increased in cover from the unburned bog to the severely burned bog. Both the unburned and moderately burned bogs had approximately 20% lichen ground cover, which was scarcely present at the unburned fen, but otherwise not present at all. At Poplar Fen, leaf litter was most prevalent at the severely burned treatment, decreasing toward the unburned treatment. The moderately burned fen had the highest percent cover of liverwort, which was only seen at the unburned fen and severely burned bog at approximately 5–10%. Both the moderately burned fen and the moderately burned bog had the greatest amount of standing water at the time of sampling. At the severely burned treatments, tree seedlings covered a higher percentage of the understory than they did at the moderately burned treatments. LMEs indicated that shrub, leaf litter, forb, and moss percent coverage was significantly different amongst sites (Table 4). A proceeding pairwise comparison (Table 5) revealed that differences in shrub percent cover were significant between the unburned bog and the severely burned bog, and the unburned bog and the unburned fen. Differences in leaf litter percent cover were found between the unburned, moderately burned, and severely burned bogs and the severely burned fen, the moderately burned bog and the moderately burned fen, and the moderately burned fen and severely burned fen. Forb percent coverage was significantly different between the moderately

burned bog and the moderately burned fen, the moderately burned bog and the unburned fen, the unburned bog and the moderately burned fen, and the moderately burned fen and severely burned fen. Finally, moss percent coverage was significantly different between the unburned bog and the severely burned fen.

According to Shannon's diversity index, the most diverse site was the severely burned bog, followed by the moderately burned bog, the unburned bog, the unburned fen, the moderately burned fen, and finally the severely burned fen, which had the lowest diversity according to Shannon's diversity index (Table 6). According to Simpson's diversity index, the severely burned bog was the most diverse, then the moderately burned bog, the unburned fen, the unburned bog, the moderately burned fen, and finally the severely burned fen had the lowest diversity according to Simpson's index. These indices differ in only the order of the unburned fen and the unburned bog.

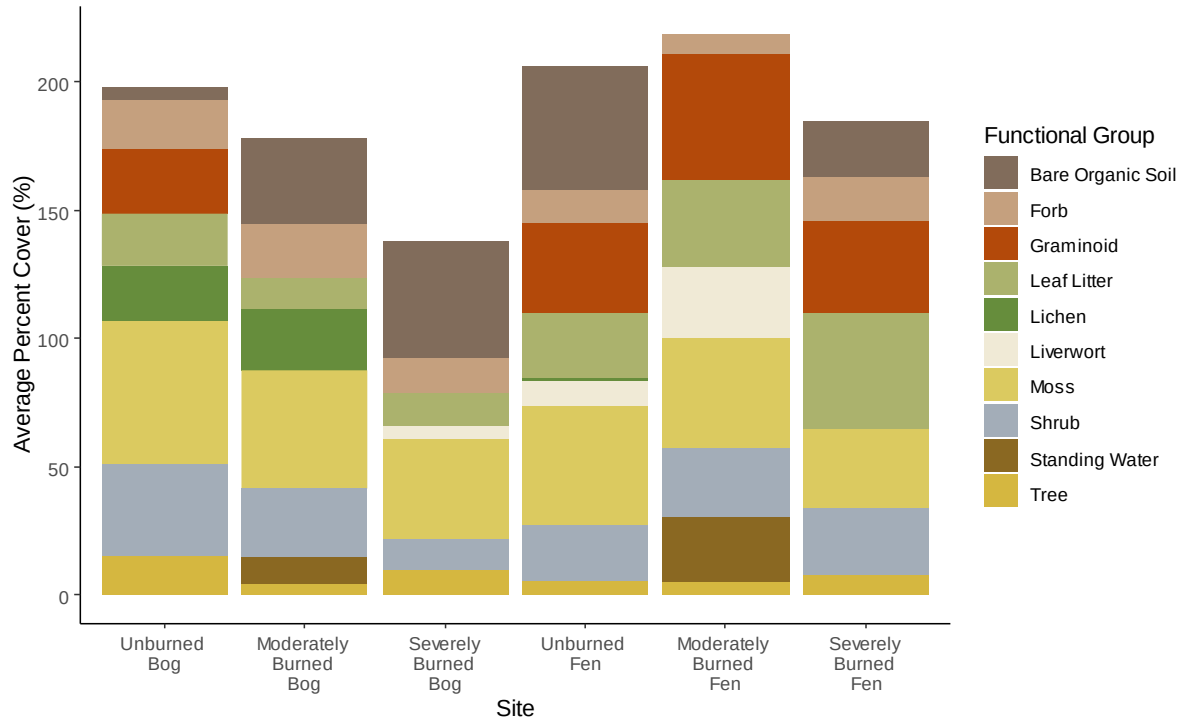


Figure 4: Bar graph depicting the variation in total aboveground biomass across all six sites. The x-axis shows the different sites sampled, while the y-axis represents grams per meter squared of biomass at each site, and the proportion of each functional group that comprises the total amount. All *Equisetum* species have been included in the forb functional group.

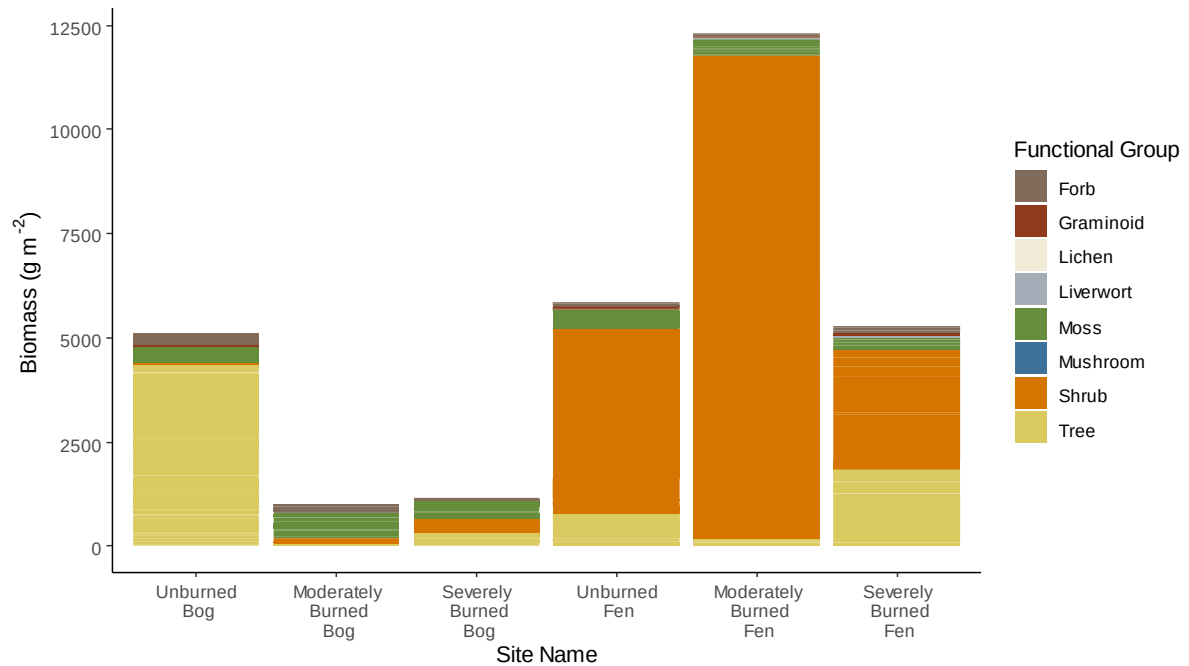


Figure 5: Bar graph depicting the variation in total aboveground biomass across all six sites. The x-axis shows the different sites sampled, while the y-axis represents grams per meter squared of biomass at each site, and the proportion of each functional group that comprises the total amount.

2.3.2 Plant functional traits

Only *R. groenlandicum* and *P. mariana* seedlings were measured across all five sites, because of variability among community compositions. Across all sites, there were no significant differences for any of the plant traits measured for *R. groenlandicum* (Figure 9), while *P. mariana* had significant differences in all traits (Figure 10; Table 2). *P. mariana* seedlings were significantly taller at the moderately burned fen than at the severely burned and unburned fen and unburned bog (Figure 6). LDMC was also lower for *P. mariana* at the moderately burned fen than both unburned sites (Figure 8), while the only difference in SLA was higher values at the severely burned fen compared to the unburned fen (Figure 7).

In regard to *R. groenlandicum*, the unburned bog and unburned fen had similar strategies, using their limited resources for horizontal and vertical growth. The *R. groenlandicum* at the severely burned fen, interestingly, had the same strategy. At the moderately burned bog, plants have ample access to resources, leading to a high LDMC, which *R. groenlandicum* invested primarily into vertical growth. The moderately burned fen had the lowest LDMC across all five sites, which *R. groenlandicum* invested mostly in horizontal growth, growing shorter but wider.

In regard to *P. mariana*, as with *R. groenlandicum*, the unburned bog and unburned fen had high resource availability, shown with the highest LDMC (Figure 8). They tended to allocate these resources to growing moderately tall, with moderately sized leaves, although *P. mariana* at the bog had a higher SLA than at the fen, and *P. mariana* at the fen were taller on average than at the bog. At the moderately burned bog, *P. mariana* seedlings split their resources similarly to the unburned treatments. At the moderately burned fen, *P. mariana* spent resources on rapid expansion, becoming the tallest on average across all five sites. At the severely burned fen, resources were also limited, but more were invested in SLA than height.

At Prairie Creek Bog, the height of both *E. arvense* and *E. sylvaticum* was greater at the unburned bog than the moderately burned bog, albeit only slightly so (Figure 6). LDMC was similar for *Vaccinium vitis-idaea* but the moderately burned bog had a greater LDMC for *E. sylvaticum* (Figure 8).

At Poplar Fen, *E. fluviatile* grew taller on average at the moderately burned site, while *E. fluviatile* growing at the unburned fen and severely burned fen had similar height values (Figure 6). *B. glandulosa* was approximately twice as tall at the moderately burned fen than the unburned and severely

burned fen sites (Figure 6). In a comparison of specific leaf area across the five sites, an LME indicated that there were significant differences within each species depending on site (Figure 7; Table 2). *E. fluviatile* was more conservative in the unburned treatment and allocated more resources to rapid leaf growth with increasing burn severity (Figure 7). *B. glandulosa* allocated fewer resources to leaf growth with increasing burn severity. A comparison of LDMC indicated a significant effect of site for *E. fluviatile* between the moderately and unburned fens (Figure 8; Table 3). At the fen sites, *B. glandulosa* and *E. fluviatile* both had greater LDMC on average at the unburned fen compared to the burned treatments (Figure 8).

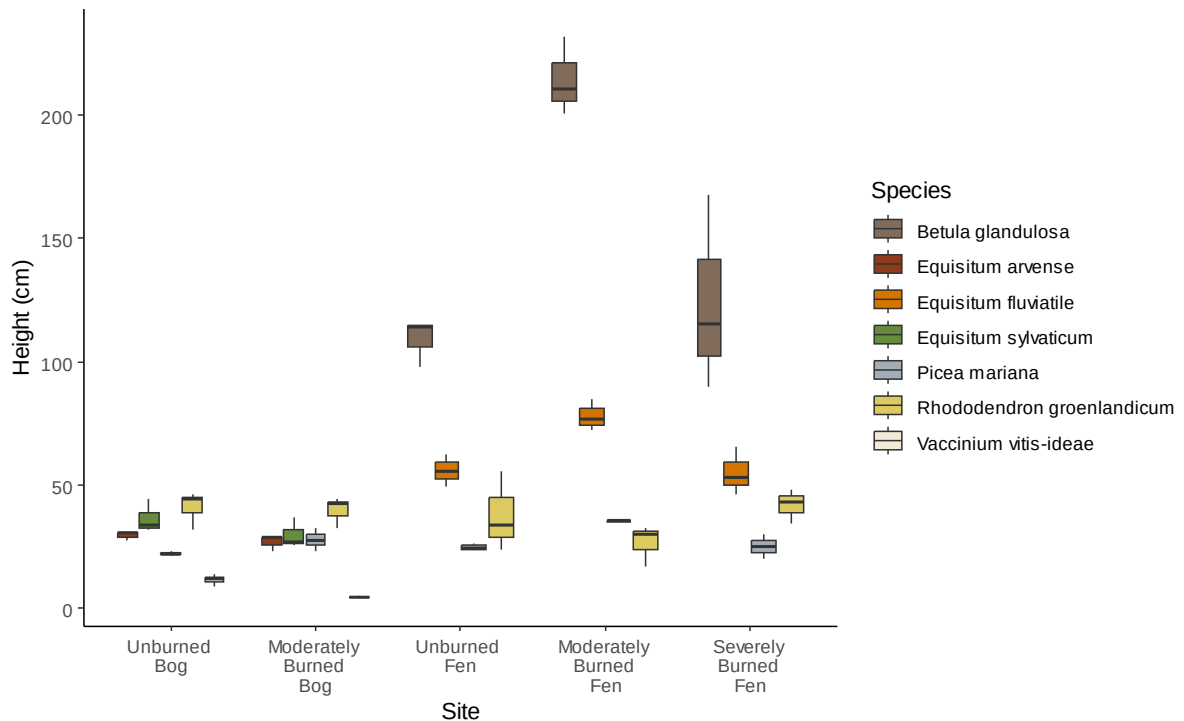


Figure 6: Boxplot depicting the variation in height (cm) across the different sites. The y-axis represents height (cm), while the x-axis shows the different sites sampled. Each boxplot represents the distribution of SLA of each species within a specific site, with the horizontal line inside the box indicating the median, the box encompassing the interquartile range (IQR), and

the whiskers extending to 1.5 times the IQR. Outliers, if present, are represented by individual points outside of the whiskers.

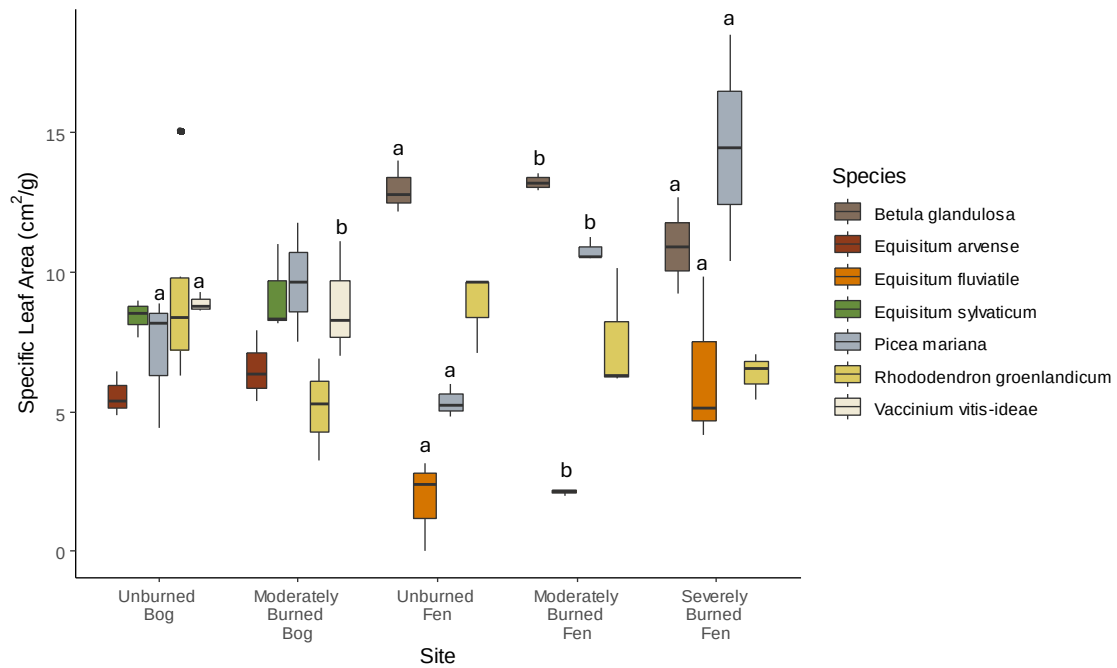


Figure 7: Boxplot depicting the variation in specific leaf area (SLA, cm²/g) across the different sites. The y-axis represents SLA, while the x-axis shows the different sites sampled. Each boxplot represents the distribution of SLA of each species within a specific site, with the horizontal line inside the box indicating the median, the box encompassing the interquartile range (IQR), and the whiskers extending to 1.5 times the IQR. Outliers, if present, are represented by individual points outside of the whiskers.

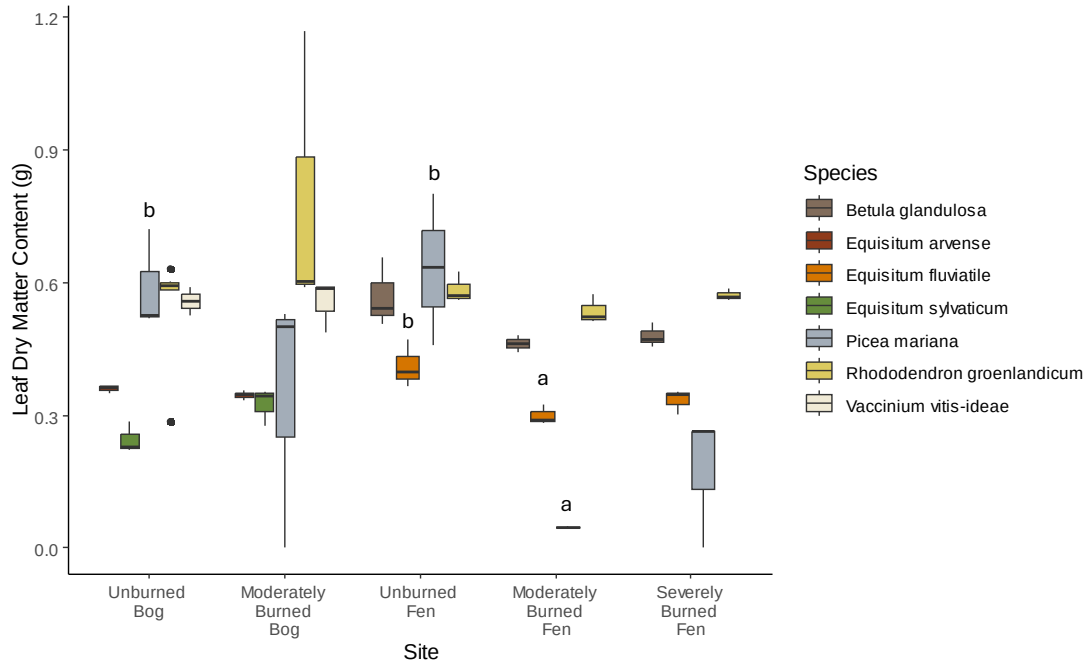


Figure 8: Boxplot depicting the variation in leaf dry matter content (LDMC, g) across the different sites. The y-axis represents LDMC, while the x-axis shows the different sites sampled. Each boxplot represents the distribution of LDMC of each species within a specific site, with the horizontal line inside the box indicating the median, the box encompassing the interquartile range (IQR), and the whiskers extending to 1.5 times the IQR. Outliers, if present, are represented by individual points outside of the whiskers.

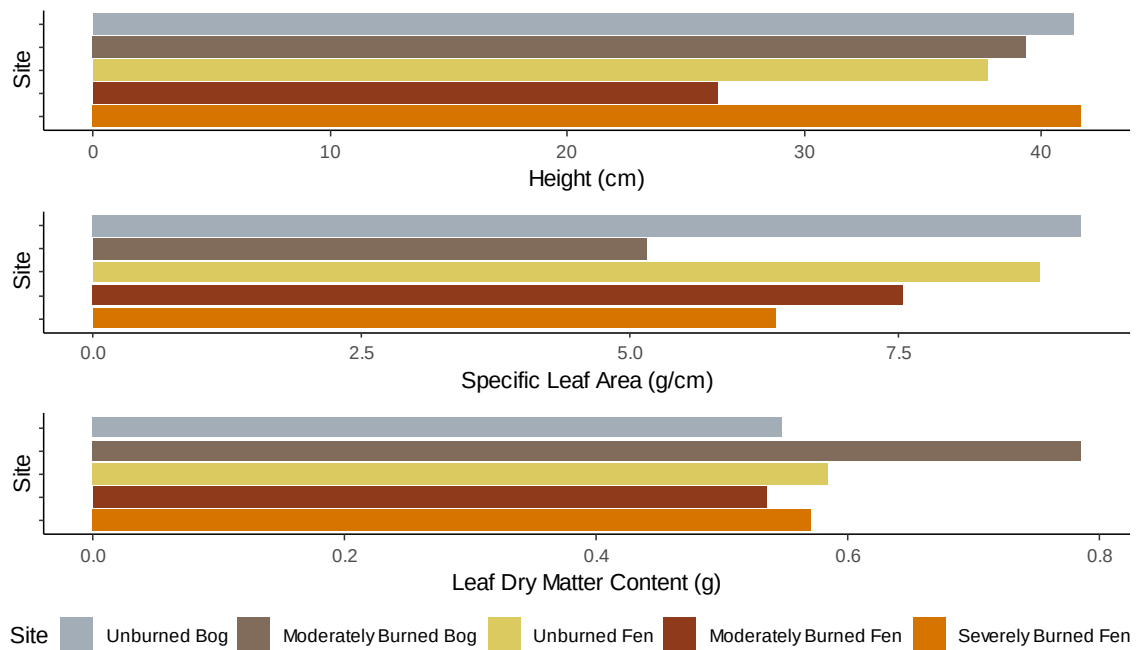


Figure 9: Bar graph depicting height (cm), specific leaf area (SLA, cm^2/g), and leaf dry matter content (LDMC, g) of *R. groenlandicum* across the different sites. The y-axis shows the five different sites (corresponding colours and names can be found in the legend), and the x-axis represents height (cm), SLA (cm^2/g), and LDMC (g) and their respective units.

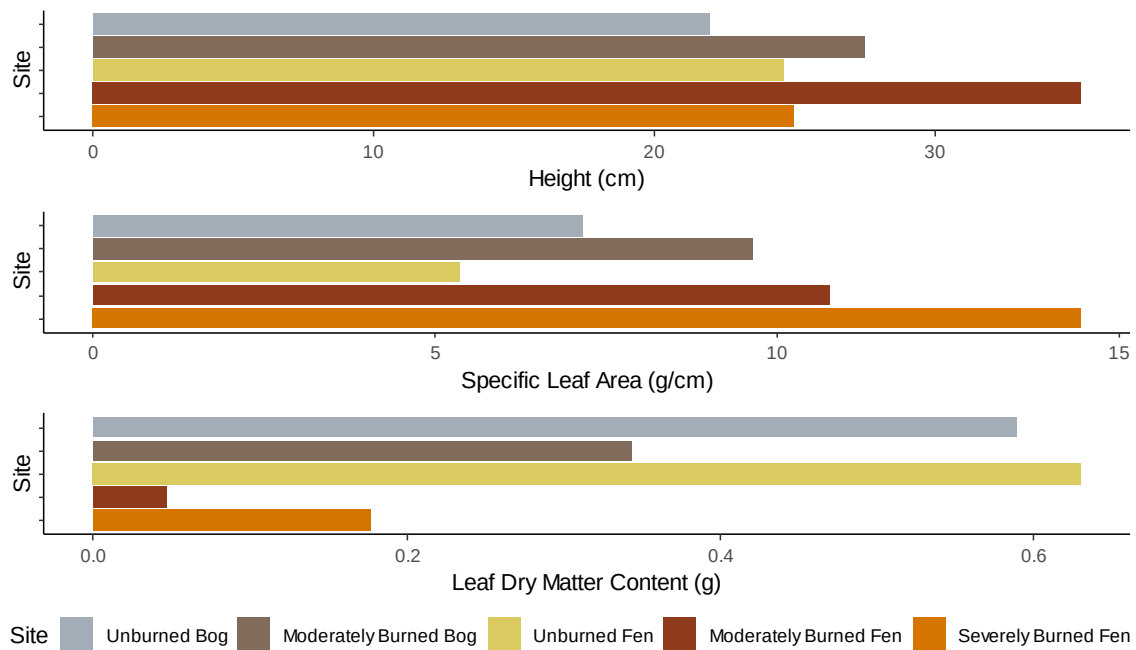


Figure 10: Bar graph depicting height (cm), specific leaf area (SLA, cm^2/g), and leaf dry matter content (LDMC, g) of *P. mariana* across the different sites. The y-axis shows the five different sites (corresponding colours and names can be found in the legend), and the x-axis represents height (cm), SLA (cm^2/g), and LDMC (g) and their respective units.

Table 2: Main effects of plant functional traits and biomass compared across sites

	Df	Sum Sq	Mean Sq	F value	Pr (>F)
R. groenlandicum					
Height	4	520.89	130.22	1.65	0.22
Specific Leaf Area	4	42.25	10.56	1.85	0.18
Leaf Dry Matter Content	4	0.14	0.03	1.43	0.28
P. mariana					
Height	4	307.70	76.92	7.13	0.01
Specific Leaf Area	4	119.33	29.83	4.38	0.04
Leaf Dry Matter Content	4	0.77	0.19	6.26	0.01
B. glandulosa					
Height	2	19614.90	9807.50	15.31	0.00
Specific Leaf Area	2	9.4	4.72	3.60	0.09
Leaf Dry Matter Content	2	0.20	0.01	4.10	0.08
E. fluviatile					
Height	2	966.47	483.23	6.71	0.04
Specific Leaf Area	2	38.67	19.34	4.85	0.06
Leaf Dry Matter Content	2	0.02	0.01	7.04	0.03
V. vitis-ideae					
Height	1	73.50	73.50	27.55	0.01
Specific Leaf Area	1	0.01	0.01	0.01	0.94
Leaf Dry Matter Content	1	0.00	0.00	0.01	0.94
E. sylvaticum					
Height	1	69.36	69.36	1.64	0.27
Specific Leaf Area	1	0.91	0.91	0.62	0.47
Leaf Dry Matter Content	1	0.01	0.01	6.18	0.07
E. arvense					
Height	1	10.67	10.67	1.54	0.28
Specific Leaf Area	1	1.41	1.41	1.25	0.33
Leaf Dry Matter Content	1	0.00	0.00	3.18	0.15
Biomass					
Site	5	76513	15302.70	1.84	0.10

Table 3: Main effect of site on percent cover of each functional group measured

	Degrees of freedom	Sum of squares	Mean square	F value	p-value
Shrub	5	5075	1014.9	2.685	0.02287
Tree	5	1137.0	227.40	2.0562	0.07827
Graminoid	3	3972	1323.88	1.9067	0.136
Leaf Litter	5	13741	2748.13	5.7377	0.0001013
Liverwort	2	725.5	362.73	0.6753	0.5248
Moss	5	11168	2233.66	2.234	0.05262
Lichen	2	452.2	226.09	0.2802	0.7593
Bare Organic Soil	4	2398.2	599.54	0.5898	0.6743
Forb	5	9358	1871.67	7.4171	1.085e-06

Table 4: Diversity indices for understory plant community along bog and fen burn severity gradients

	Unburned Bog	Moderately Burned Bog	Severely Burned Bog	Unburned Fen	Moderately Burned Fen	Severely Burned Fen
Simpson's diversity index	0.937	0.930	0.895	0.936	0.938	0.944
Shannon's diversity index	2.940	2.869	2.402	2.950	2.955	3.012

2.4 Discussion

2.4.1 Vegetation community composition

Analysis of vegetation community composition elucidates the potential responses of bogs and fens to fire across a burn severity gradient. Our results indicate that in the moderately burned fen, liverwort cover was greater than in the unburned fen. However, Davies et al. (2023) observed a decrease in liverwort percent cover with increasing burn severity. Previous work (Hobbs 1986) has also suggested that liverworts may be sensitive to fire, which contrasts with our findings. The distribution of forbs exhibits notable trends across the different burn treatments and ecosystems. Forb abundance was highest in the unburned bog and decreased progressively towards the severely burned bog. Grau-

Andrés et al. (2019) found that along a fire severity gradient in a raised bog, presence of forbs increased, opposing the results of this study, as forbs decreased under the same conditions. In the same study, Grau-Andrés et al. (2019), noted moss percent cover decreasing along the burn severity gradient, in accordance with the results demonstrated in this study (Figure 4). Conversely, at Poplar Fen, forb abundance is lowest in the unburned treatment and increased towards the severely burned treatment. Once again, these results differ from the literature, where Shepherd et al. (2021) found no significant interactions between burning and habitat type, which suggests that wildfires had similar impacts on species regardless of the ecosystem type where measurements took place. Guéné-Nanchen et al. (2022) found that along a burn severity gradient forbs were smaller in stature than they were at their unburned treatment. This is also contradictive of the findings of this study, where forbs, and plants on average, grew larger along the burn gradient. As increased access to light and nutrients became available, I posit that plants took advantage, and grew larger, faster, to outcompete others during a period of rapid succession. In contrast, within fens, forb abundance is lowest in unburned areas and increases with burn severity, diverging from Shepherd et al. (2021), who found no significant interactions between burn severity and habitat type, suggesting uniform wildfire impacts on species across different ecosystem types. At each severely burned site, there is greater tree biomass than the moderately burned sites, likely a result of opportunistic species taking advantage of the ideal conditions created by the moderate burn. Conversely, the severe burn allowed *P. mariana* seedlings to establish without competition. *P. mariana* has been shown to have increased seedling establishment under traditional fire return intervals, but when the interval between fires is decreased, seedling establishment is also decreased. Further, old growth in ecosystems that have not been recently burned prevents seedling establishment due to a reduction in the substrate quality (Brown and Johnstone 2012). Our study indicates that species recovery varies among peatland classes, with distinct recovery trajectories for each type.

There are important distinctions that may account for disparities between the literature and our study. Firstly, all three referenced studies were conducted in Eurasia, often under controlled conditions. Even in uncontrolled settings, Eurasian fire regimes differ significantly from North American boreal fire regimes (Rogers et al. 2015). Consequently, the burn severity gradients in this study are markedly different from those in other studies, which often define the most severe fires based on amount of burned vegetation, and not involving soil combustion. In contrast, the burn severity classifications in this study were based on the depth of burn, leading to crucially different and significant distinctions from the existing literature.

As well as community composition, peatland types also vary in the response of biomass six to seven years post fire. At Prairie Creek Bog, biomass declined, largely due to a loss of trees. At Poplar Fen, biomass was similar between the severely burned and unburned, and greatest at the moderately burned fen. This is due to shrub and tree biomass recovery and results due to faster growing species like *B. glandulosa* and *L. laricina* at the fen. In a study of fire vegetation recovery in open fens, Guêné-Nanchen et al. (2022) noted increased presence of shrubby species including *B. glandulosa* and *Salix brachycarpa*. It has been suggested that some species could return to a burned site faster than others as a result of their reproductive strategy being more favourable to fast regeneration such as the use of rhizomes or resprouting, possibly even enhancing its total biomass following disturbance.

According to the diversity indices, the severely burned bog is the most diverse. In this context, diversity—comprising species richness and species evenness—may indicate resource availability, with increased access to light, nutrients, and water enabling plants to establish before being outcompeted. In the unburned bog, as an established peatland, few new species can establish due to shading and competition from older vegetation for water and nutrients. While Shepherd et al. (2021) noted a general

trend of decreasing diversity with increasing burn severity, Davies et al. (2023) observed increasing diversity and species richness in bog communities, aligning with our findings. Disturbance often promotes diversity, as pre-disturbance species return and opportunistic species exploit the increased availability of resources, (Lukenbach et al. 2016; Davies et al. 2023), although this relationship is closely dependent on the frequency of disturbance (Enright et al. 2014). At Poplar Fen, the opposite trend is observed; the unburned fen exhibits the highest diversity compared to moderately and severely burned treatments. It is likely that the severe burn affected the seedbank and soil microbial community, limiting the potential for peatland plants to return post-fire. In fact, Shepherd et al. (2021) observed that even at a post-fire site with no burned soil, the seed bank was reduced to approximately half of the species that were present aboveground pre-fire. Furthermore, Ward et al. (2012) reported a significantly decreased microbial load following wildfire. In the moderately burned fen, which shows lower diversity indices compared to the unburned fen, a significant proportion of the community is composed of the shrub *B. glandulosa*. It is possible that, instead of increasing diversity, *B. glandulosa* capitalized on the increased resource availability and outcompeted slower-growing vegetation.

2.4.2 Plant functional traits

During wildfire recovery, plants may favour resource acquisitive strategies of growth given the increase in light and nutrient availability (Limpens et al. 2008; Pekin et al. 2011; Laine et al. 2021). In this case, one would expect a reduction in SLA and LDMC and an increase in plant height at burned sites compared to unburned sites. This is true of the moderately burned bog and moderately burned fen, where plants were taller on average compared to their respective unburned treatments (Figure 6). These results indicate that resource acquisitive strategies are favoured by most plants following disturbance. *P. mariana* and *R. groenlandicum* appear to use the same strategies, both having lower LDMC and

higher SLA post-fire (Figures 9 & 10). Although they are not necessarily taller, these plants grew larger leaves, with less associated C accumulation. Thus, although resources are ample, they may be responding to increased access to light by growing larger leaves. The *R. groenlandicum* at the severely burned fen, interestingly, uses the same strategy. This may indicate a reduction in labile nutrients and a reduced soil microbial community, which under pre-fire (normal) conditions would be capable of acquiring nutrients as part of a symbiotic relationship. At the moderately burned bog, plants had ample access to resources, leading to a high LDMC in *R. groenlandicum*, which was invested mostly in vertical growth. The moderately burned fen has the lowest LDMC across all five sites, which *R. groenlandicum* invested mostly in horizontal growth, opting to grow slightly shorter but with broader leaves. *P. mariana* at the unburned bog and unburned fen had high resource availability, as indicated by the greatest LDMC. They tended to allocate these resources to moderate vertical and horizontal growth compared to the same plant at the burned treatments, although *P. mariana* at the bog had a higher SLA than at the fen, and *P. mariana* at the fen were taller on average than at the bog. In these conditions, *P. mariana* seedlings can grow rapidly, which is a benefit to plants in the early stages of succession, as plants carve out a niche for themselves within the site, outcompeting others for access to nutrients, light, and water. Although there are no measurements of plant functional traits at the severely burned bog, some informed hypotheses can be made based on community composition compared to the other bog sites, and comparisons to trends in the severely burned fen data. It appears severe burns, rather than allowing for new growth opportunity, cause retrogression of a site, likely burning the seed bank (Kelly et al. 2016), the soil microbial community (Mikita-Barbato et al. 2015), and making drastic changes to integral peatland functioning such as the water table and the soil temperature. These factors reduce the rate of recovery.

2.5 Conclusion

Peatlands exhibit varied recovery patterns following wildfire. Conflating multiple peatland classes into a single monolithic group may contribute to substantial errors in estimating recovery trajectories and C accumulation. Bogs and fens show markedly different responses in terms of biomass recovery: fens accumulate the same amount of biomass or more six to seven years post-fire, whereas bogs accumulate significantly less. This disparity may be due to the acidic nature of bogs, which restricts tree and shrub growth, while in fens, trees and shrubs return and grow more rapidly. Similarly, conflating microtopographic forms into a single group may mask internal variations in recovery (Bubier et al. 1993); however, microtopographic forms were not assessed during the data collection in this study, and provide an avenue for future research.

Some of the reported findings contrast with the few studies that have investigated post-fire recovery in the context of plant functional traits. These discrepancies may be explained by several factors.

First, most research on vegetation response to wildfire is based on field studies conducted in Eurasia, which has different fire regimes compared to North America (Rogers et al. 2015). There are studies conducted in North America, but they tend to address post-fire vegetation community composition rather than plant functional traits (e.g. Benscoter and Vitt 2008; Lukenbach et al. 2016). Second, the depth of burn in controlled or even uncontrolled fires in Eurasia does not match the extensive soil combustion observed in Canadian boreal fires. These differences make comparisons challenging, underscoring the need for more studies focused on Canadian ecosystems to better understand vegetation recovery trajectories in peatlands following wildfire.

However, this study is not infallible, there are limitations and uncertainties that may have affected the outcomes. First, there were no pre-fire measurements, making the assignment of difference among the

sites to the effect of fire alone difficult. Comparisons were made on the assumption that the landscape was the same across the burn severity treatments prior to the wildfire, however without pre-burn site information it is not possible to be certain. Also, restricted by time and resources, this study included only two sites. Data from these sites have been applied broadly based on ecosystem type; however, this cannot be the case outside of context, and without reference to other factors such as local climate, which would present differences in fluxes.

Plant functional traits are a practical metric for measuring differences in plant community resource allocation strategies. In this study, plant functional traits were used to assess differences in plant life strategies along a fire severity gradient, and between two peatland types, an ombrotrophic bog, and a moderate-rich fen. The results of the study suggest that plants used a more rapid accumulation strategy along the fire severity gradient as nutrients became labile and access to light, water, and growing space was increased. This trend was the case between the moderately burned and unburned sites, however when assessing the impact of a severe burn, the pattern halted. It appears that as the severe burn went deeper into the peat, the favourable impact of greater light and nutrient availability was overwhelmed by the disruption to the seed bank and soil microbiome.

Chapter 3

Post-fire CO₂ and CH₄ fluxes as an indicator of recovery in boreal peatlands

3.1 Introduction

Global climate change is increasing the frequency and severity of wildfires in the boreal region (Turetsky et al. 2015), burning stored organic matter and releasing its carbon (C) into the atmosphere (Lasslop et al. 2019). This is of augmented importance in peatlands, where C has been accumulating for millennia (Yu et al. 2010). This release of stored C as greenhouse gases (GHGs) contributes to global climate change, which further affects northern peatlands through increased severity and frequency of drought and higher soil temperatures, making them more susceptible to wildfires (Turetsky et al. 2011). Although contemporary research has provided an understanding of the importance of these disturbance events, they have not yet been successfully included in global C cycle models, underrepresenting the true efflux of C from the Earth to the atmosphere (Hashimoto et al. 2004; Eskelinen and Harrison 2015; Rogers et al. 2015; Santín et al. 2016; Van Der Werf et al. 2017; Hu et al. 2018; Wilkinson et al. 2018; Laine et al. 2019; Lasslop et al. 2019). Quantitative data are needed to better understand the impact of peatland fires on the global C cycle, as well as wildfire return intervals based on post-fire conditions. This study aims to quantify recovery of C exchange in peatlands affected by wildfire.

Peatlands are ecosystems distinguished by their exceptional capacity to store C in soils and biomass, due to inputs of dead organic matter exceeding the rate of decomposition (Clymo 1984). Peatlands occupy only 2–3% of the Earth's total area but store 25–33% of the total global soil C stock (Gorham 1991; Turetsky et al. 2014; Van Der Werf et al. 2017). In Canada alone, northern boreal peatlands occupy only 12% of the total land area but store 50% of the country's terrestrial C stock

(Tarnocai 2006). Peat is maintained in these ecosystems by the presence of a typically high water table, which creates anoxic conditions, and prevents rapid C oxidation that typically occurs during decomposition in dryland ecosystems. Microtopographic features in peatlands consist of raised mounds termed hummocks, and hollows, which are depressions in the surface topography and result in heterogeneous water table depths across the peatland. Under normal conditions (wet) hummocks tend to be much drier than hollows, and as a result, each microform has preferential plant species associations which propagate and enforce differences in the water balance (Sumner 2007). When disturbed by fire, the ecohydrological functioning integral to peatland stability is disrupted.

During fires in peatlands, aboveground biomass is burned, and depending on severity, some belowground biomass and organic soil (Benscoter and Wieder 2003; Turetsky et al. 2011; Brown et al. 2015; Lukenbach et al. 2016; Grau-Andrés et al. 2019; Davies et al. 2023). Peat does not burn evenly across peatland microforms, as fires tend to preferentially burn hollows (Benscoter and Wieder 2003), which under drought conditions are unable to retain water near the surface as effectively as hummocks. Thus, fires tend to propagate into hollow microforms, since the increased thermal energy necessary to dry out hummocks to use as fuel is energetically unfavourable. Depending on the severity of the burn, hummocks may be retained, or as is often the case in severe burning, hummocks will tend to become closer in elevation to hollows (Benscoter and Wieder 2003). With severe burning, more above and below ground biomass and peat are lost to the atmosphere as GHG fluxes (French et al. 2002; Davidson et al. 2019). Fire severity alters surficial soil properties, increasing hydrophobicity and bulk density (Thompson and Waddington 2013); radiative flux, increasing exposure to direct sunlight (Brown et al. 2015; Li et al. 2017); hydrologic cycling, with lower canopy interception and evaporative losses from the soil (McCarter and Price 2014); and soil microbial communities, decreasing microbial biomass

(Orwin and Ostle 2012; Turetsky et al. 2014). In these ways, burn severity alters peatland functioning, but few studies have assessed the influence of burn severity on the timing of recovery (Nelson et al. 2021).

Under historical climatic conditions, peatlands have been able to recover from burning on a decadal timescale, re-sequestering the C lost to fire via rapid biomass accumulation (Turetsky et al. 2011). However, under a warming climate, peatlands are experiencing drier conditions, and without the high water table that has typically conferred deep peat resilience to burning (Kettridge et al. 2015), fires are burning deep into stocks of peat that have been stored for millennia. The effects of fire on the immediate release of C are well-understood (French et al. 2002; Ballhorn et al. 2009; Turetsky et al. 2015; Kohlenberg et al. 2018; Hu et al. 2019); however, the understanding of C flux dynamics during the recovery period following disturbance is still limited. Current studies of peatlands recovery from wildfires suggest that within 10 years peatlands will have returned to C sinks (Wieder et al. 2009), but these studies fail to consider the impact of fire severity on the recovery trajectory. Studies often also fail to differentiate between peatland classifications, and tend to study only ombrotrophic bogs, although there are a few studies on moderate rich fens (e.g. Davidson et al. 2019; Van Beest et al. 2019; Morison et al. 2021), because of this, little is known about how different peatland classifications recover following wildfire.

Therefore, the objective of this study is to quantify CO₂ and CH₄ fluxes along microtopographic gradients from a bog and fen, located in north-eastern Alberta, across a fire severity gradient. The aim is to gain a better understanding of wildfire recovery trajectories, which can inform future Earth systems models. There are two main hypotheses (1) seven years following a wildfire, moderately burned sites would be further along the recovery trajectory than the severely burned sites, meaning they would be

more similar to the unburned treatments in terms of ground level C fluxes, and (2) that the bog would be further along the recovery trajectory than the fen in terms of C fluxes.

3.2 Methods

3.2.1 Site location and Description

The study was conducted in a moderate-rich treed fen (hereafter referred to as Poplar Fen; 56° 56' N, 111° 32' W) and an ombrotrophic bog (hereafter referred to as Prairie Creek Bog; 56° 57' N, 111° 30' W). Prairie Creek Bog and Poplar Fen are located approximately 31 km south and 29 km north, respectively, of Fort McMurray, Alberta. Both sites were affected by the 2016 Horse River wildfire and exhibit varying burn severities across each site. From May to August, the mean temperature in 2022 was 15 °C, and the mean total precipitation during the same period was 58 mm. During the same months in 2023, the mean temperature was 17 °C, and the mean total precipitation was 73 mm. Over the past decade, the mean temperature from May to August was 15 °C, and the mean total precipitation was 66 mm (Extracted from the Environment and Climate Change Canada Historical Climate Data web site https://climate.weather.gc.ca/index_e.html on December 18, 2023).

We identified and instrumented portions of each study site representing three burn severities, severely burned with 14.5-17 cm of burned peat at the fen, and 4 cm at the bog, moderately burned with 9-11 cm of burned peat at the fen and 1 cm at the bog, and unburned, with 0 cm of burned peat. These burn severities, although vastly different, are based on the respective depths observed in each peatland complex, and so are representative of the “severe” and “moderate” burns within each site. Once burn severities had been established, paired hummock/hollow collars were installed to account for differences in microtopography, since it has been shown that microforms have different C fluxes

(Moore et al. 2011) and depth of burn. For example, (Benscoter and Wieder 2003) observed higher depth of burn in hollows due to greater vulnerability of hollow species to fire during drought conditions. As a result, hummock/hollow pairs may not have looked the same in each of the burn severities and were based on assumptions of pre-fire microtopography. In 2022, Poplar Fen was instrumented with four pairs of collars at each burn severity, 24 total. At each burn severity, 4 collars were placed in hummocks, and 4 collars were placed in adjacent hollows. In the same year, Prairie Creek Bog was instrumented with three pairs of collars at the unburned site, and three pairs of collars at the moderately burned site, with three placed in hummocks, and three in their adjacent hollows. During the field season of 2023, more collars were added, so that the design of Prairie Creek Bog was the same as Poplar Fen; four pairs of collars at each burn severity, with 48 total collars across the two study sites. Each collar is approximately 20 cm in diameter and between 15-30 cm in height. Upon installation each collar was hammered into the ground as far as possible, with effort taken to make the collar flush with the soil or moss surface.

3.2.2 CO₂ flux

To measure ground layer CO₂ fluxes, the closed chamber method was used following Alm et al. (1997). Chambers had dimensions of $d = 20$ cm, $h = 40$ cm resulting in a volume of approximately 12.5 L. To measure total volume, the difference between the collar rim and the ground were measured, and volume was corrected based on this measurement. To create a seal between the ground and the chamber, water was added to the rim of established collars. During the measurement period, a battery-operated fan located at the top of the chamber was used to mix the chamber headspace, preventing a buildup of gas at the top of the chamber. Chambers were also instrumented with a thermocouple wire and a photosynthetic active radiation (PAR) sensor, so that during CO₂ flux measurements, PAR, and air

temperature in the chamber headspace were also recorded. At the beginning of the measurement period, chambers were vented to bring the internal gas composition and temperature to approximately ambient, and then placed into the rim of the collar. The concentration of CO₂ in the chamber headspace was measured with an EGM-4 Infrared Gas Analyzer (PP Systems, Massachusetts, USA) and recorded every 15 seconds for 1.5–2 minutes. CO₂ fluxes were calculated from the linear change in CO₂ concentration over time and converted to units of CO₂ m⁻² d⁻¹ using the ideal gas law, molar mass of CO₂, chamber volume corrected for temperature, and collar surface area. Full light flux measurements were considered net ecosystem exchange (NEE), where a negative value represents net uptake of CO₂ by the ecosystem. To measure only ecosystem respiration (ER), an opaque neoprene sleeve was used to cover the chamber, preventing any light from reaching plants. Gross ecosystem productivity (GEP) was then calculated as the difference between NEE and ER. All values were inspected for linearity and those with R² < 0.90 were rejected except when the change in concentration over the closure period was within the precision of vial measurements (~0.5 ppm) as these represent near zero fluxes. These quality control procedures resulted in the loss of 4.2% of the data.

3.2.3 CH₄ flux

To measure CH₄ flux, the closed chamber method was also used (Alm et al. 1997), wherein a sealed chamber is placed at ground level, and CH₄ fluxes are measured for 25 minutes, with physical air samples taken at 5, 10, 15, and 25 minutes. To create a seal between the ground and the chamber, water was added to the rim of established collars, creating a water-tight seal. Dark chambers are used throughout the sampling period to reduce solar heating of the chamber during the closure period. During the time of measurement, a battery-operated fan located at the top of the chamber was used to mix the chamber headspace, preventing a buildup of gas at the top of the chamber. Chambers were also

instrumented with a thermocouple wire to monitor chamber headspace temperature during closure. Physical samples were taken through a tube connected to the chamber headspace that was sealed between measurements to prevent external and internal air mixing. 20 mL samples were taken at each time point and injected into 12mL pre-evacuated Exetainers. Concentration of CH₄ in these samples was determined in the laboratory using a Fast Methane Analyser (FMA) (Los Gatos Research DLT-100, ABB Ltd., Zürich, Switzerland). Once the samples had been injected into the FMA alongside standard concentrations of CH₄ at 1 ppm, 5 ppm, and 50 ppm, the program (PEDRO Newton (2023). PEDRO - Peatland Equipment Data Re-organizer (Version 1.0.0) [Computer software] <https://github.com/BrianNewton/PEDRO>) was used to calculate the area under the curve from the injection and compare it to the standard CH₄ concentrations. CH₄ flux was calculated from the linear change in CH₄ concentration in the chamber over time. All values were inspected for linearity and those with $R^2 < 0.75$ were rejected except when the change in concentration over the closure period was within the precision of vial measurements (~0.5 ppm) as these represent near zero fluxes. Fluxes where the 5-minute CH₄ concentration was greater than 5 ppm followed by a decline in concentration over time were also rejected, as these likely represent a release in CH₄ during chamber placement followed by diffusion back into the soil. These quality control procedures resulted in loss of 31% of the data.

3.2.4 Collar vegetation composition and greenness

Vegetation within collars was identified to the species level using Johnson et al. (2017) and quantified visually into percent cover, so that the total vegetation in each collar may be greater than 100%.

The greenness of the vegetation in the collars was monitored throughout the sampling period using the green chromatic coordinate (GCC) following the methods outlined in (Davidson et al. 2021).

Downward facing photos of each collar were taken alongside each flux measurement. Photos were taken at a height of approximately 30 cm, or the least distance required to capture the entire collar within the width of the lens when held in the portrait direction. The R package *Phenopix* version 2.3.1 (Filippa et al. 2016) was used to calculate GCC, where R_{DN} , G_{DN} , and B_{DN} are the average red, green and blue digital numbers, respectively.

$$GCC = \frac{G_{DN}}{R_{DN} + G_{DN} + B_{DN}}$$

The *DrawROI* function was used to outline the Region-Of-Interest for each plot within the boundaries of the flux collar. The *ExtractVIs* function then calculated the chromatic coordinate indices on a per-pixel basis (Toomey et al. 2015). Collar estimates are based on an average of all greenness indices collected at a collar over the growing season.

3.2.5 Environmental variables

Site characteristics including pH and EC were measured (Table 5) using an Oakton™ Thermo Scientific PCSTestr 35 (Oakton, Cole-Palmer, IL, USA). pH and EC were measured by squeezing the water out of the surface peat near each collar into a clean plastic container, submerging the probe and waiting for the reading to equilibrate. Measurements were made on June 27, 2023, and June 30, 2023, with no significant rainfall between measurements.

Table 5: pH and EC values from each site

Site	pH	EC (mS/cm)
Unburned Bog	5.60	189
Moderately Burned Bog	6.04	188

Severely Burned Bog	4.85	235
Unburned Fen	4.94	233
Moderately Burned Fen	4.58	196
Severely Burned Fen	5.41	369

Water table was measured in 1.5-m long, 2.5 cm diameter PVC pipes inserted 1 m into the soil adjacent to each collar. Installation was completed as early in the season as possible, although soil frost layers prevented early installation at some sites. Collar elevations were measured in comparison to the water table wells, and so water table compared to the surface at each collar during flux measurements could be determined.

Soil temperature was measured during each C flux measurement at 30 cm, 20 cm, 15 cm, 10 cm, 5 cm, and 2 cm depths using a 30 cm Digi-Sense WD-93758-02 thermocouple immersion probe (Cole-palmer instrument company, Illinois, USA) and TME MM2020 Dual Input Thermocouple Thermometer (TME, West Sussex, UK).

3.2.6 Data analysis

All data were analyzed using R version 4.2.2 (R core Team, 2022). To assess the impact of wildfire or environmental variables on site, linear mixed effects models (LME) were run using the function ‘lme’ in the package *nlme* (Pinheiro and Bates 2000). Water table was run with the fixed effects of site, microform, soil temperature at 5 cm and soil temperature at 30 cm. The random effect of collar was included to account for multiple sampling at each collar over the growing season. Models for soil temperature at 5 cm and 30 cm, GEP, ER, NEE, and CH₄ flux were all run with the fixed effects of site and microform, and the random effect of collar. To separate the impact of environmental variables from treatment on fluxes, linear mixed effects models were created for GEP, ER, NEE, and CH₄ flux with the fixed effects of soil temperature (30 cm for CH₄, as these flux values are often more closely related

to changes at or below the water table, and 5 cm for NEE, GEP, and ER, as these flux values are found to be more closely related to the area above the water table), water table, and site, and the random effect of collar. If any of the fixed effects were found to be significant, pairwise comparisons were evaluated using the function ‘emmeans’ in the package *emmeans* (Lenth R, 2023) with a Tukey adjustment.

To assess the effect of fire on GCC, a linear mixed effect model was created using the function ‘lme’ in the package *nlme* (Pinheiro and Bates 2000). Site, microform, and their interaction were fixed effects, and collar was included as a random intercept to account for repeated measures at each collar. The LME was run through a type-3 ANOVA to determine which variables contributed to variance within the means of the data. The ‘emmeans’ function in the *emmeans* (Lenth R, 2023) package was then used for a pairwise comparison if the outcome of the ANOVA was a p-value >0.05 for site, which had a Tukey adjustment to account for multiple testing at collars.

3.3 Results

3.3.1 Environmental variables

To determine if there were any environmental differences that had impacted the rate of C fluxes, total monthly precipitation and average monthly temperature in each year of study was compared, provided by historical weather data from the Fort McMurray International Airport (https://climate.weather.gc.ca/index_e.html) (Figure 11). Overall, precipitation was more consistent over the summer months of 2022, while in 2023 precipitation was far higher in June than in other summer months. In both 2022 and 2023, average temperatures peaked at approximately 18 °C (Figure 11). In 2022 the temperature rose at a steady rate until peaking in July and August, and then began to steadily decrease as the growing season ended. In 2023, temperatures were within 4 °C of one another

from May through to September. Compared to the 30-year climate data (Figure 11), precipitation in 2022 followed similar patterns and amounts, while precipitation in 2023 was far greater and with less evenly distributed, predominantly occurring in June. Temperature patterns in 2022 and 2023 were similar to the long-term average temperatures.

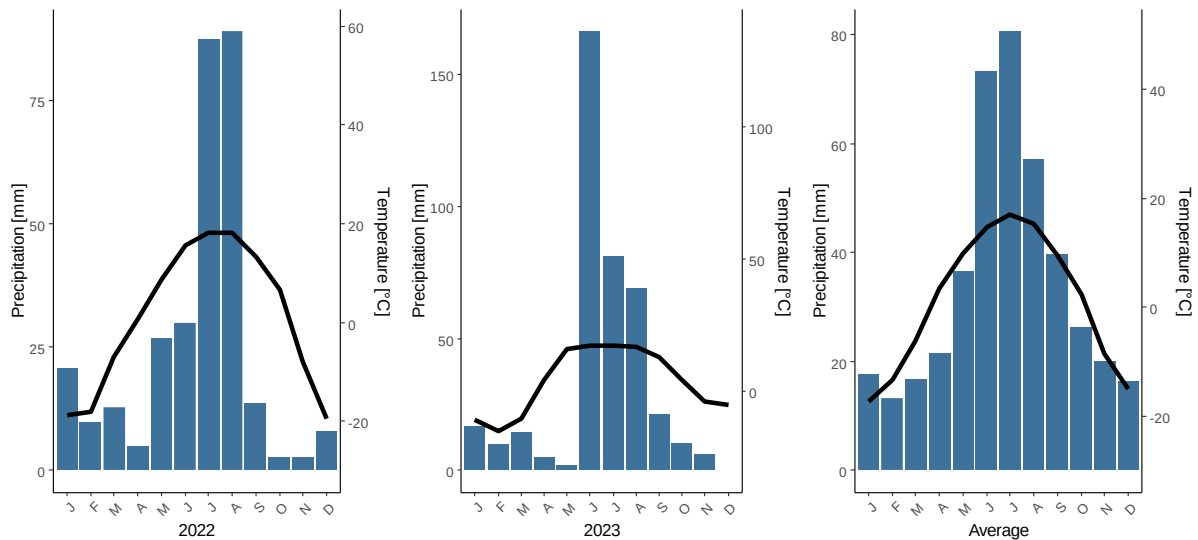


Figure 11: Fort McMurray climographs for 2022, 2023, and 30-year average climate data measured at the Fort McMurray International Airport. Temperature is represented by the black line and the scale on the right side of the graph. Precipitation is represented by bars, each bar showing total precipitation measured over the respective month, using the scale on the left side of the graph. Note that scales are variable, and as such the data portrayed must be considered according to its respective scale.

Across all study sites, the water table depth ranged from 89 cm below the moss surface to 37 cm above the moss surface, representing the maximum and minimum values observed across the entire study area (Figure 12). A linear mixed effects model (LME) indicated significant difference between

microforms, but not sites, and no significant interaction between the two variables (Table 6). The hummocks tended to have a greater range of water table depths than hollows, which on average have a water table nearer the ground surface. The range between minimum and maximum water table depths in hummocks decreased with burn severity, with the unburned bog and fen having a larger range of observed values than the moderately burned bog and fen, decreasing again for the severely burned bog and fen, whose microforms had greater similarity in water table depth, indicating a dampening of the hydrological variation between microforms following fire.

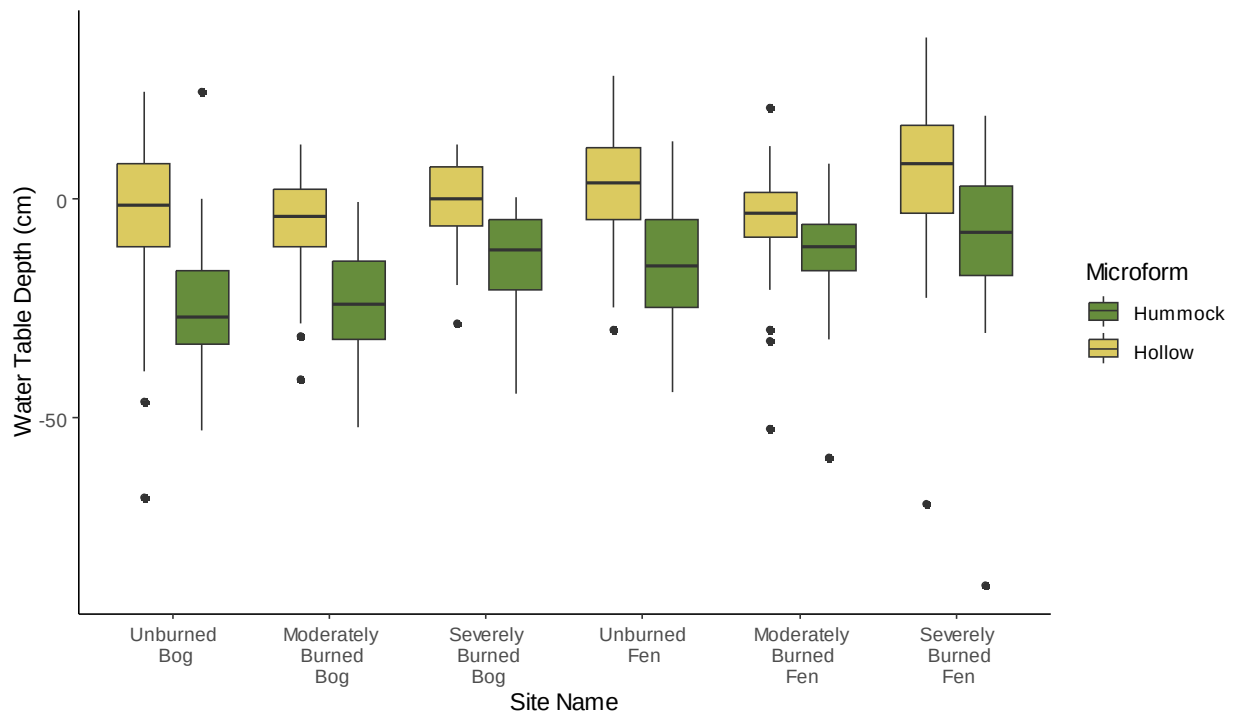


Figure 12: Boxplot depicting the variation in water table levels across different sites. The y-axis represents the water table depth in centimeters (cm), while the x-axis shows the different sites sampled. Each boxplot represents the distribution of water table depths within a specific site,

with the horizontal line inside the box indicating the median depth, the box encompassing the interquartile range (IQR), and the whiskers extending to 1.5 times the IQR. Outliers, if present, are represented by individual points outside the whiskers.

Across both peatland classes, average soil temperature at 5 cm depth ranged from a maximum of 39.4 °C to a minimum of 1 °C, any temperatures lower than 1 °C were frozen ground and were therefore not possible to measure with the equipment used (Figure 13). Hummocks had consistently higher 5 cm soil temperature than hollows, with slightly warmer temperatures at moderately and severely burned sites compared to unburned. Results from the LME model indicated no significant interaction between site and microform, but there were significantly warmer soils at hummocks than hollows, and a significant effect of site (Table 6). The severely burned fen had a significantly higher 5 cm soil temperature than all other sites, which were statistically similar based on pairwise comparisons (Table A 3).

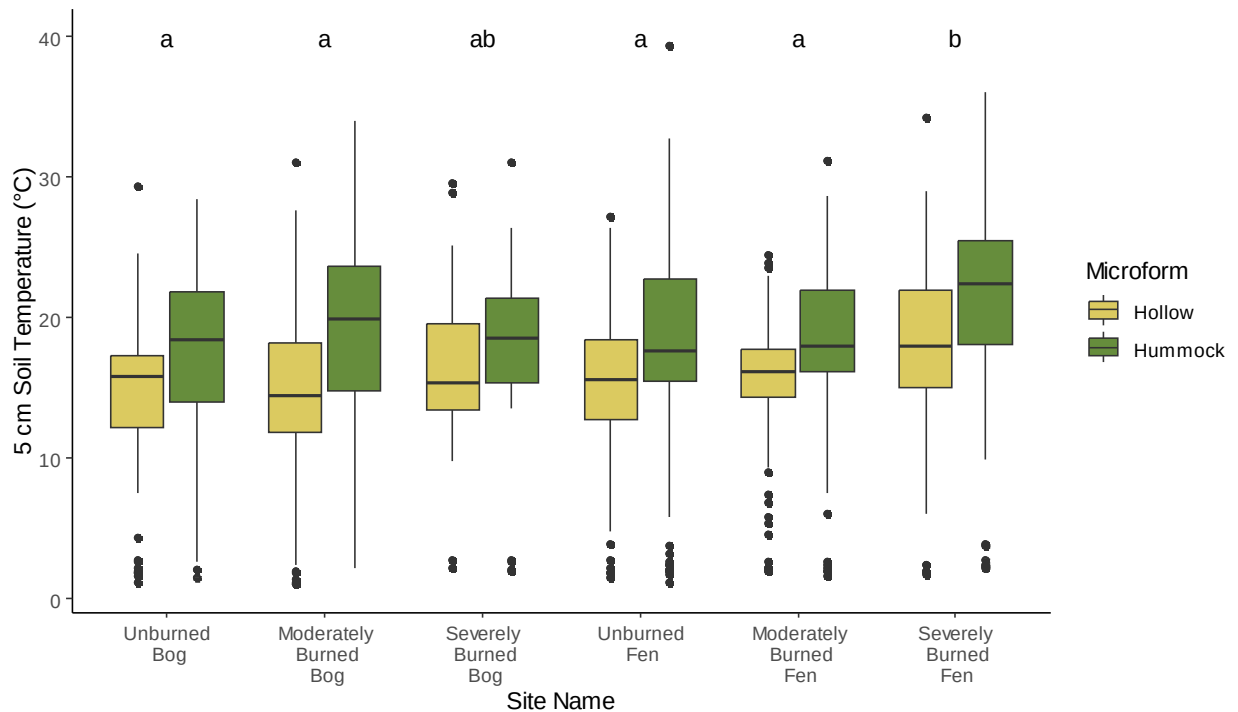


Figure 13: Boxplot depicting the variation in near surface soil temperature (5 cm) across different sites. The y-axis represents the soil temperature in degrees Celsius (°C), while the x-axis shows the different sites sampled. Each boxplot represents the distribution of 5 cm soil temperatures within a specific site, with the horizontal line inside the box indicating the median depth, the box encompassing the interquartile range (IQR), and the whiskers extending to 1.5 times the IQR. Outliers, if present, are represented by individual points outside the whiskers.

At 30 cm depth, soil temperatures followed a similar pattern to those measured at 5 cm, albeit cooler and more muted (Figure 14). There was a significant effect of site, microform, and the interaction between site and microform (Table 6). Amongst the bog sites, the severely burned bog had significantly higher 30 cm soil temperatures than the unburned and moderately burned treatments. At the bog, both

hummocks and hollows increased in temperature with increasing burn severity, the hollows having a greater increase among treatments. At the fen, the unburned treatment was significantly different from the moderately and severely burned treatments, and the hummocks behave the same as they do in the bog sites (Table A 3). The hollows follow a different pattern in the fen than in the bog, where average temperatures increase from the unburned to moderately burned fen and decrease from the moderately burned fen to the severely burned fen. There was no significant relationship between water table and soil temperature at 5 or 30 cm depths. However, in general soil temperatures were cooler during wetter conditions (Figure A 1 & Figure A 2)

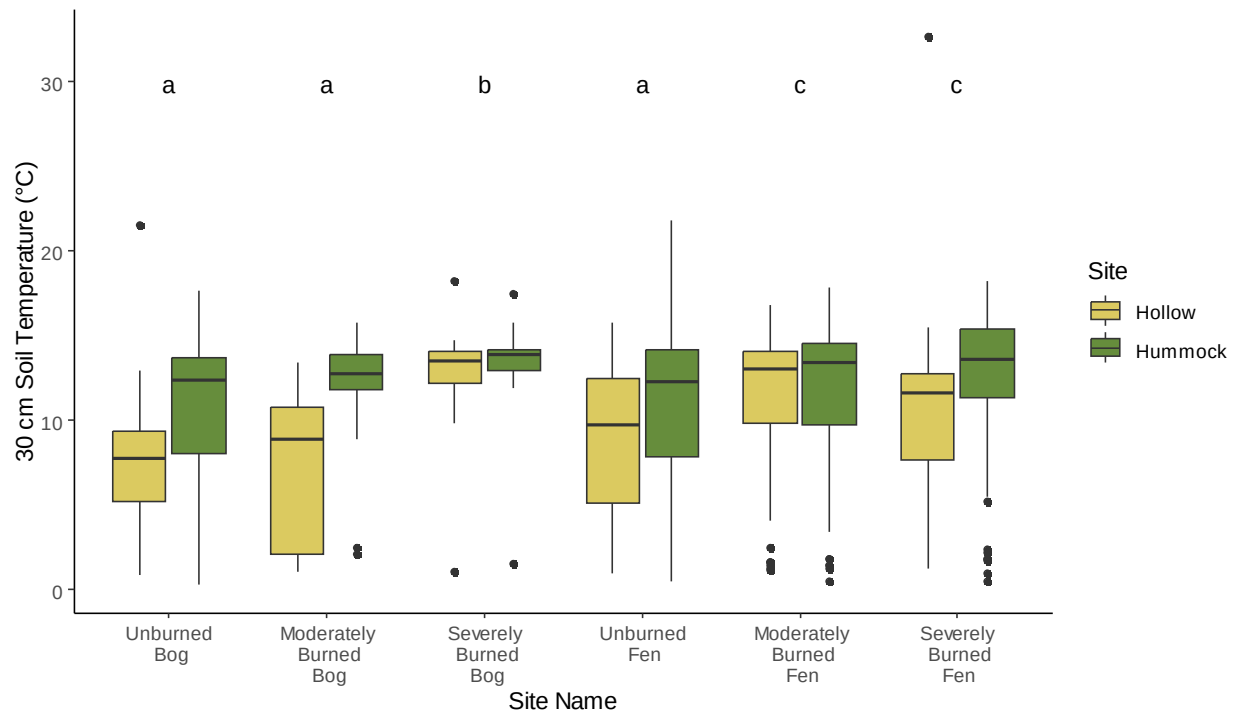


Figure 14: Boxplot depicting the variation in 30 cm soil temperatures across different sites. The y-axis represents the 30 cm soil temperature in degrees Celsius (°C) while the x-axis shows the different sites sampled. Each boxplot represents the distribution of 30 cm soil temperatures (°C) within a specific site, with the horizontal line inside the box indicating the median depth, the box encompassing the interquartile range (IQR), and the whiskers extending to 1.5 times the IQR. Outliers, if present, are represented by individual points outside the whiskers.

3.3.2 Plant community composition and greenness

In a comparison of green chromatic coordinate (GCC) at all sites, the minimum greenness index value calculated was 0.29, and the maximum was 0.47 on a 0-1 scale. Although an LME indicated no significant effect of microform, or the interaction between site and microform (Table 6), typically hollows had a greater range in values over the growing season at the bog, and hummocks had a greater range in values over the growing season at the fen (Figure 15). The model revealed a significant effect of site, where for the most part, the bog and fen were significantly different with higher GCC at the fen. The average GCC values at the bog decreased from unburned to severely burned, but at the fen, the severely burned site had the highest mean greenness index, then the unburned, followed by the moderately burned.

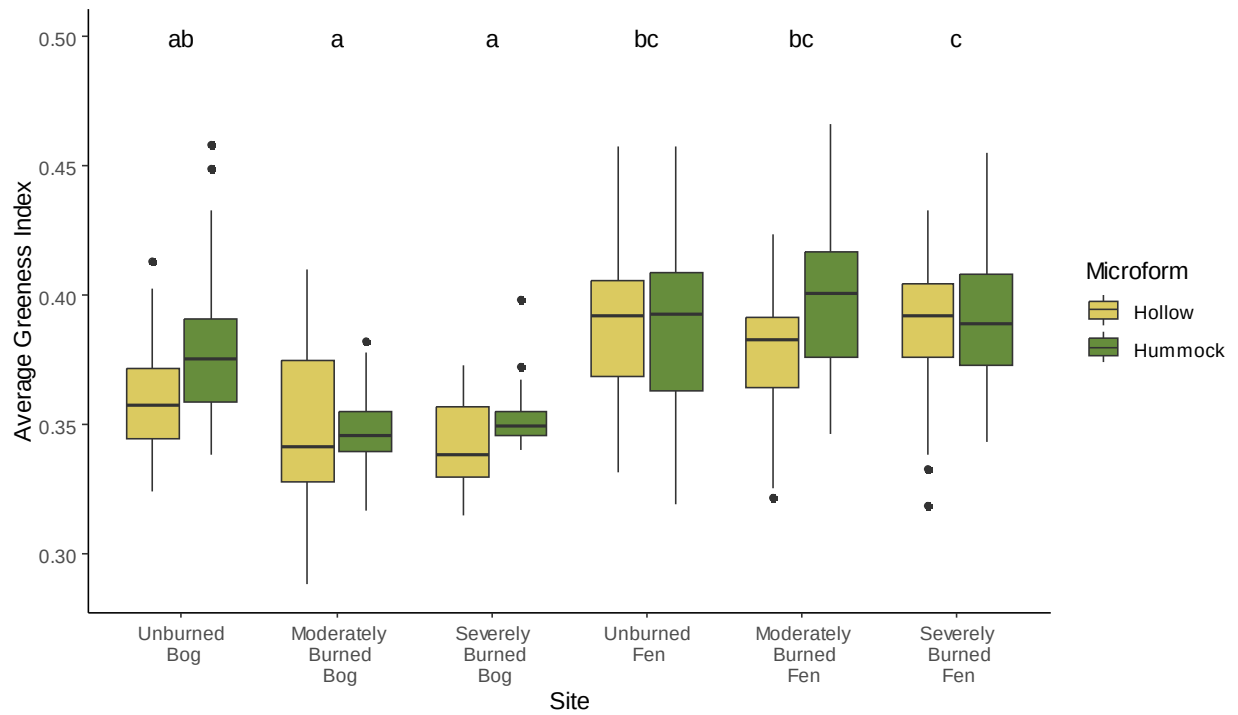


Figure 15: Boxplot depicting the variation in Green Chromatic Coordinate (GCC) across different sites. The y-axis represents GCC while the x-axis shows the different sites sampled. Each boxplot represents the distribution of GCC values within a specific site, with the horizontal line inside the box indicating the median depth, the box encompassing the interquartile range (IQR), and the whiskers extending to 1.5 times the IQR. Outliers, if present, are represented by individual points outside the whiskers.

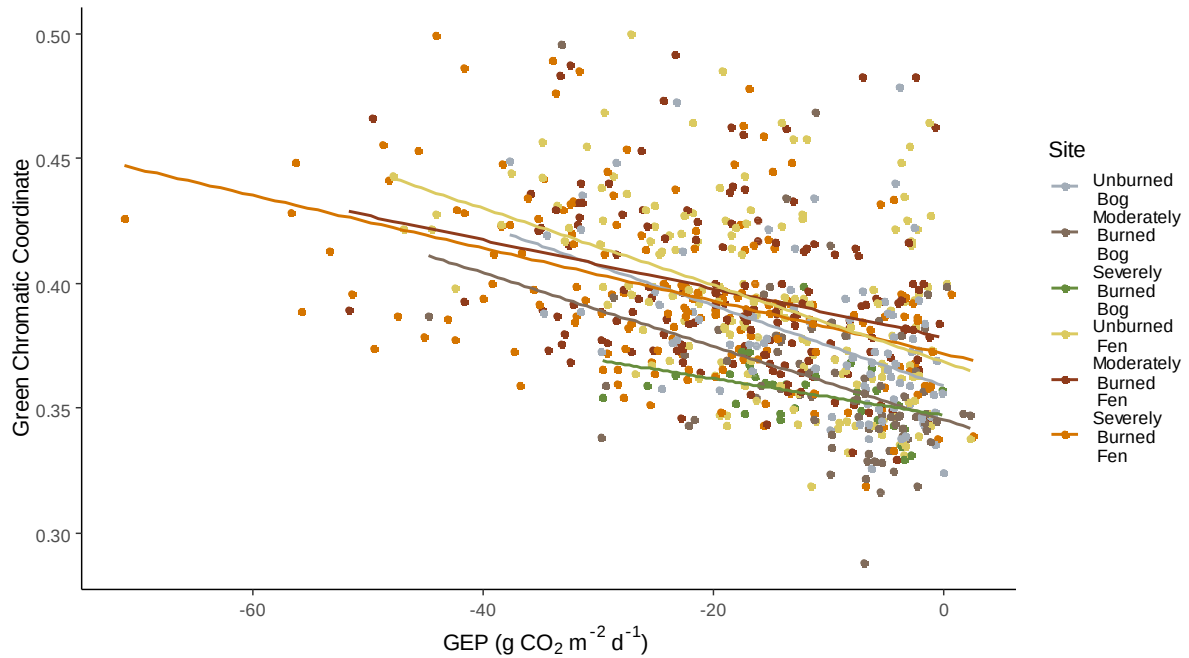


Figure 16: Scatterplot depicting the relationship between gross ecosystem productivity (GEP), and green chromatic coordinate (GCC). The x-axis represents GEP and the y-axis represents GCC.

Vascular vegetation and ground percent cover surveys of individual collars were used as an estimate of site-specific vegetation composition. There were no significant effects of site or microform, or their interaction on total vegetation cover or ground percent cover in the collars (Table 9). Bare organic soil cover was lowest at the unburned bog, increased at the moderately burned bog, and again at the severely burned bog (Figure 17). In contrast, the unburned bog has nearly twice as much moss as the burned treatments. This pattern can also be noted in the shrub percent cover, the unburned bog has a higher percent cover on average than the moderately and severely burned bogs. The unburned fen compared to its burned treatments has the highest percent cover of bare organic soil. The unburned and severely burned fens have a similar community composition, while the moderately burned fen has a

greater cover of liverwort and shrubs. Further details on the plant community composition are reported in Chapter 2.

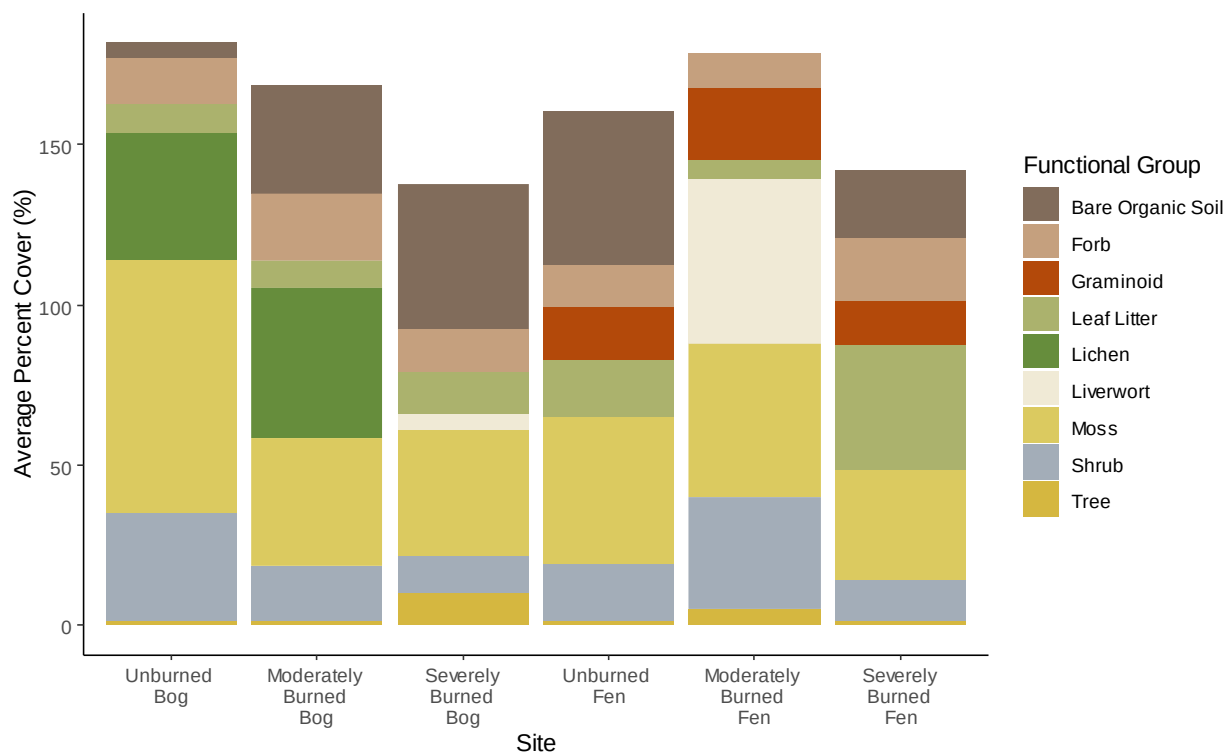


Figure 17: Bar graph depicting the average percent cover measured from all eight collars at each site. The y-axis represents average percentage (%), while the x-axis shows the different sites sampled.

3.3.3 Carbon gas exchange

The fen understory was more productive than the bog (lower GEP, greater C uptake), with slightly greater productivity at the severely burned fen site (Figure 18). GEP values ranged from 2.4 g CO₂ m⁻²

$\text{m}^2 \text{ day}^{-1}$ to $-71.2 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$. In a comparison of all six site types, an LME model indicated significantly different GEP values between sites, but not between microforms, or the interaction of site and microform (Table 6), this datum explained only 15% of the variance within the model, while 26% was explained by the fixed and random effects combined (Table 11). GEP at the fen was significantly lower (more negative, greater C uptake) than the bog, and the severely burned fen had significantly lower GEP values than all other sites. There was no significant difference in GEP along the burned severity gradient at the bog. All fen sites also had a wider range of GEP values (~ 0 to $-50 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$), with bog sites remaining more consistent over the growing season (~ 0 and $-30 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$). Within the fen sites, the hummocks had greater productivity with increasing burn severity. This relationship is also notable in the bog hollows where productivity increased with burn severity, but not significantly.

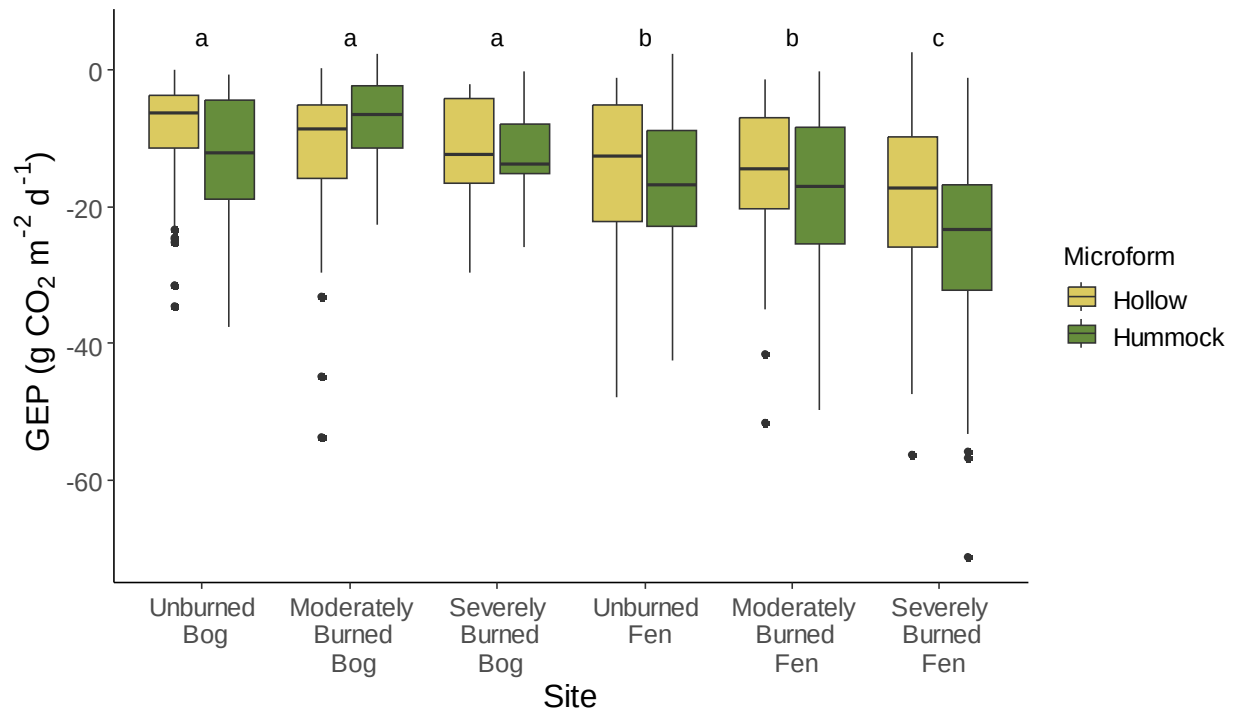


Figure 18: Boxplot depicting the variation in Gross Ecosystem Productivity (GEP) across different sites. The y-axis represents GEP while the x-axis shows the different sites sampled. Each boxplot represents the distribution of GEP values within a specific site, with the horizontal line inside the box indicating the median depth, the box encompassing the interquartile range (IQR), and the whiskers extending to 1.5 times the IQR. Outliers, if present, are represented by individual points outside the whiskers.

While water table alone did not significantly explain variation in GEP across all sites, the interaction between water table and site was significant, suggesting that GEP fluxes respond to variations in the water table differently depending on the site in which measurements are taken (Table 6; Figure 18). The three-way interaction between 30 cm soil temperature, water table, and site was also

significant, indicating that GEP fluxes responded to variations in the water table and 30 cm soil temperature, depending on the site in which measurements took place. At the moderately burned and severely burned bog sites, as well as the unburned fen, GEP decreased with an increasing water table (closer to the moss surface; Figure 19). The opposite was true of the severely burned fen, moderately burned fen, and unburned bog, where GEP decreased with increasing water table.

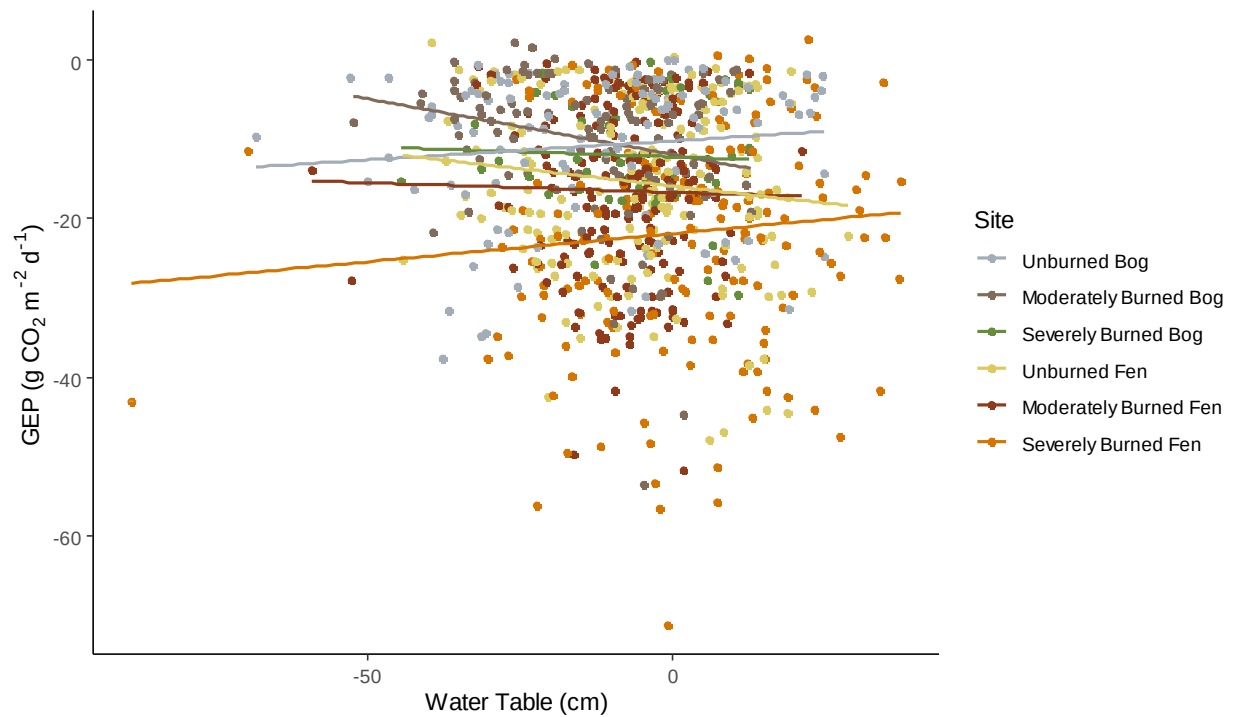


Figure 19: Scatterplot depicting the relationship between Gross Ecosystem Productivity (GEP) and water table at each site. The y-axis represents GEP, while the x-axis represents water table depth (cm).

Neither 5 cm soil temperatures, the interaction between 5 cm soil temperature and site, or the interaction between 5 cm soil temperature and water table were significant (Table 6). GEP flux measurements and soil temperature had a negative relationship across all six sites where productivity was greater at higher soil temperatures (Figure 20). The slope of this relationship is steeper at the moderately and severely burned treatments of the fen sites, and the severely burned treatment at the bog sites.

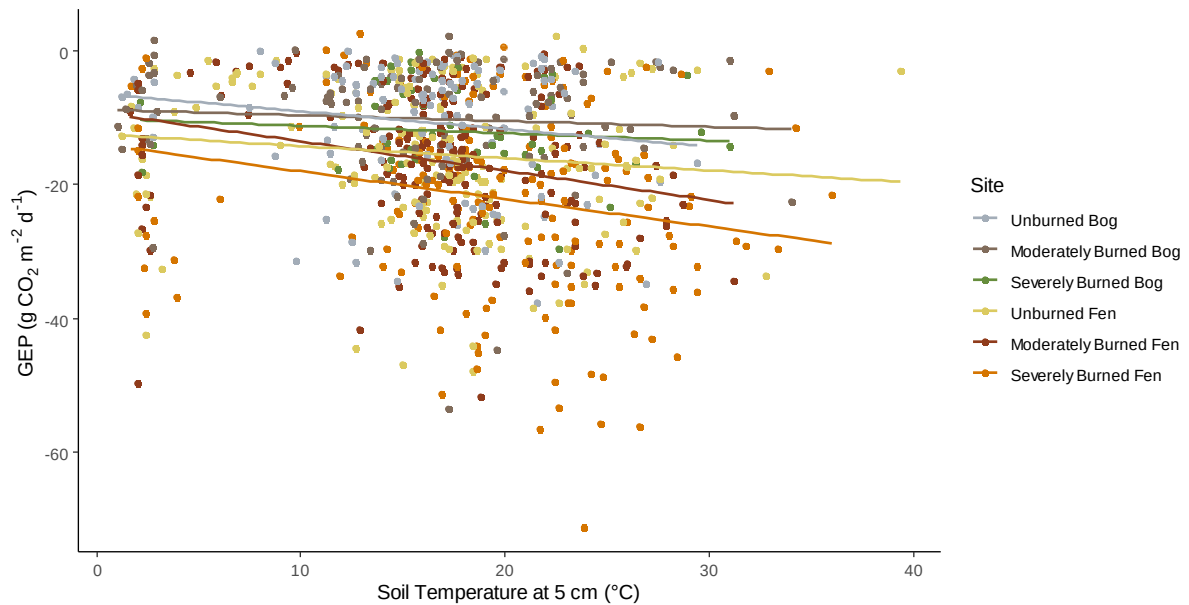


Figure 20: Scatterplot depicting the relationship between Gross Ecosystem Productivity (GEP) and 5 cm soil temperature at each site. The y-axis represents GEP, while the x-axis represents 5 cm soil temperature (°C).

Across all six sites, ecosystem respiration (ER) ranged from a minimum of $0.82 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$ to a maximum of $53.9 \text{ g CO}_2 \text{ m}^{-2} \text{ day}^{-1}$, which was observed at the unburned fen site (Figure 21). The ER LME model indicated significant differences between sites where, as observed for GEP, ER was significantly greater at all fen sites compared to all bog sites (Table A 3). Although the LME model indicated no significant effect of microform, or the interaction between site and microform (Table 6), hollows had consistently higher ER than hummocks at each site, other than unburned and moderately burned fen, where the hummocks were greater sources of CO_2 via ground level respiration than the hollows.

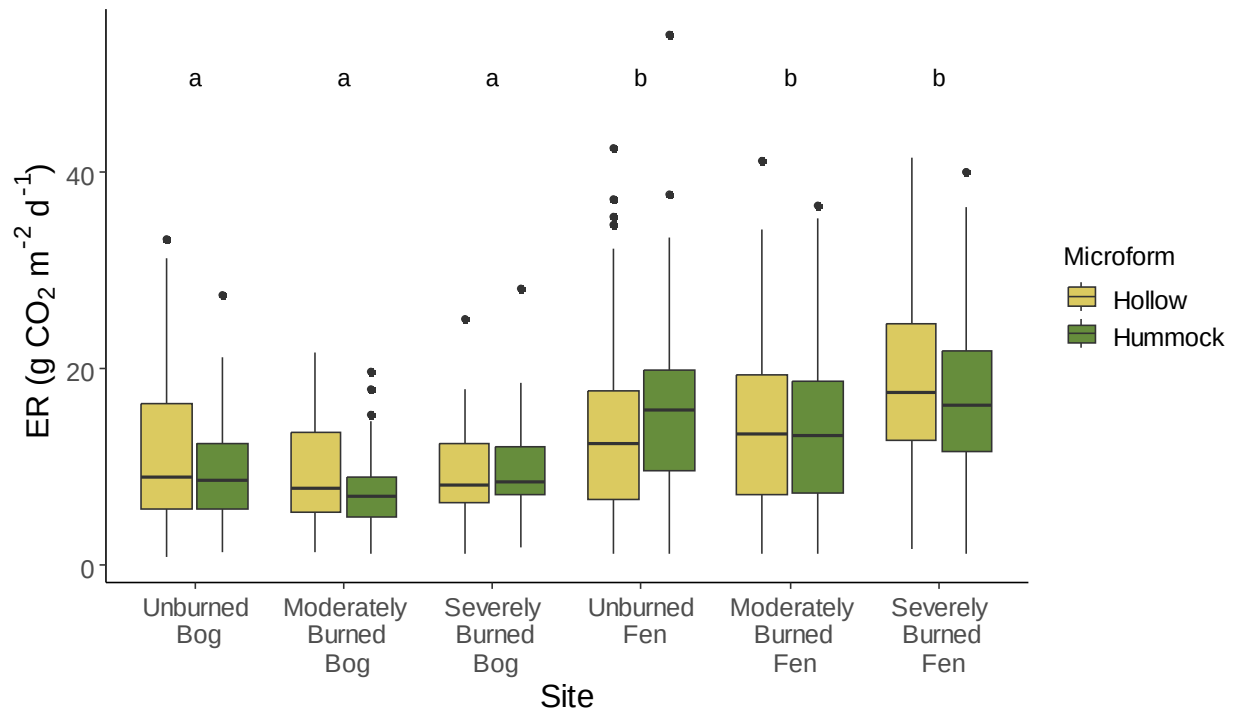


Figure 21: Boxplot depicting the variation in Ecosystem Respiration (ER) across different sites. The y-axis represents ER while the x-axis shows the different sites sampled. Each boxplot represents the distribution of ER values within a specific site, with the horizontal line inside the box indicating the median depth, the box encompassing the interquartile range (IQR), and the whiskers extending to 1.5 times the IQR. Outliers, if present, are represented by individual points outside the whiskers.

The model which included water table, 5 cm soil temperature, site and their interactions explained 33% of the variation in ER, while the fixed effects themselves explained only about 18% of the variation in ER (Table 11). Within this model, there was a significant effect of site and the interactions between 5 cm soil temperature and site, water table and site, and 5 cm soil temperature and

water table and site, while 5 cm soil temperature, water table, and the interaction between 5 cm soil temperature and water table were non-significant (Table 6). The significant interactions suggest that ER responds differently to changes in the water table depending on the site where measurements took place (Figure 22). The same principle applied to 5 cm soil temperature measurements. The three-way interaction between water table, 5 cm soil temperature, and site indicates that ER flux measurements respond differently to changes in the water table and changes in 5 cm soil temperatures depending on the site where measurements were taken. These interactions can be confounding and challenging to tease apart but indicate that the increase in ER with temperature varies for a given water table depth and that these responses also vary among sites. Ecosystem respiration and water table had a positive relationship at all bog treatments, as well as the moderately burned fen (Figure 22). The unburned and severely burned fens, however, have a negative relationship between an increase in water table (closer to the moss surface), and a decrease in ER (greater values indicate greater release of CO₂).

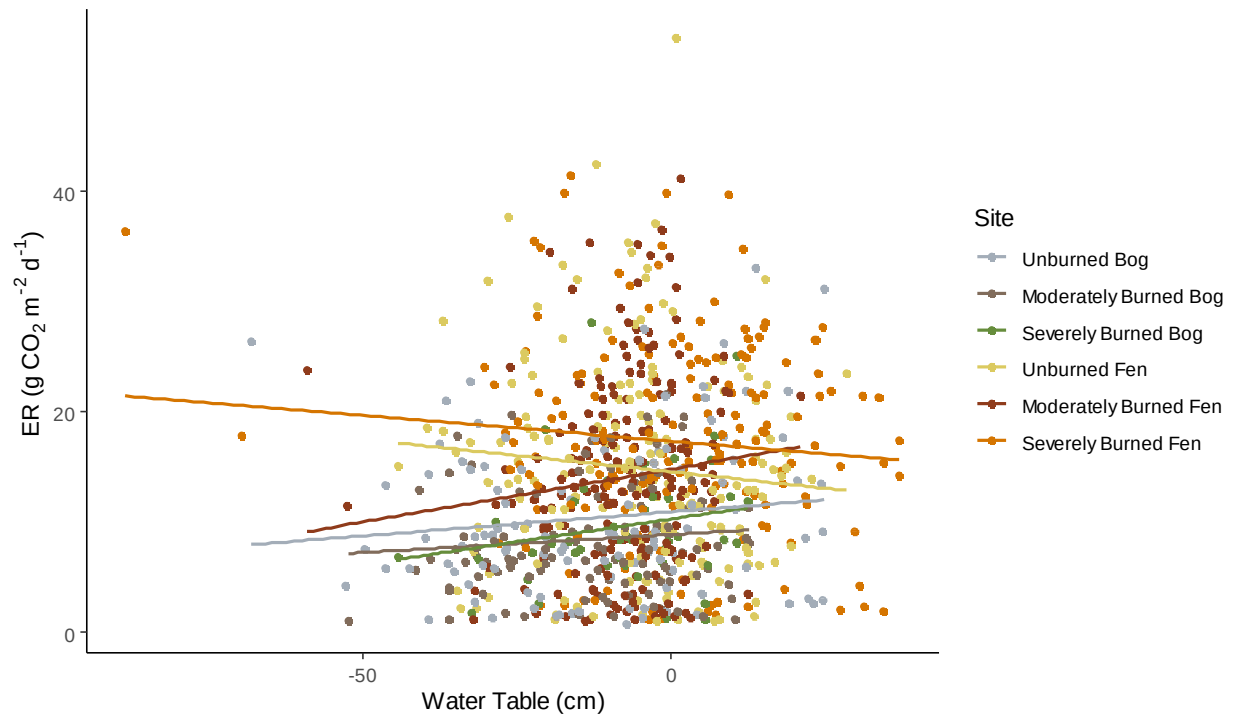


Figure 22: Scatterplot depicting the variation in Ecosystem Respiration (ER) across water table depths (cm). The y-axis represents ER while the x-axis represents water table depths (cm).

Ecosystem respiration and 5 cm soil temperature were dependent on one another, and this relationship was significantly different among sites with a significant interaction between 5 cm soil temperature and site (Table 6). Soil temperature at 5 cm and ER were positively related at all sites other than the moderately burned fen, where an increase in soil temperature was related to a decrease in respiration (Figure 23). At the moderately burned fen, an increase in 5 cm soil temperature was related to a decrease in ER. Ecosystem respiration at the unburned fen was the site most strongly positively associated with 5 cm soil temperature, while the moderately burned fen was the site most strongly negatively associated with 5 cm soil temperature. The bog has a similar relationship with temperature

and ER to the fen, however in the case of the bog, the unburned site had the strongest positive relationship between ER and near-surface soil temperature, the severely burned bog slightly less so, and the moderately burned bog had the most negative association between soil temperature at 5 cm and ER.

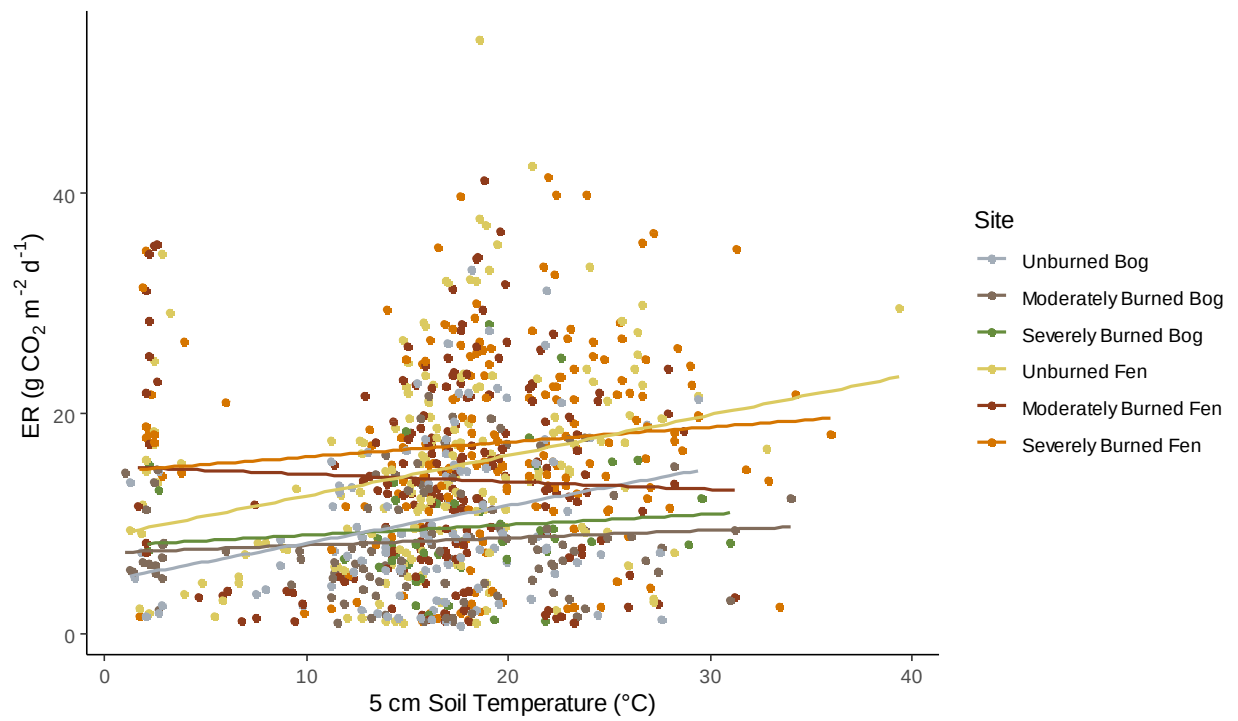


Figure 23: Scatterplot depicting the variation in Ecosystem Respiration (ER) across 5 cm soil temperatures (°C). The y-axis represents ER, while the x-axis represents 5 cm soil temperatures (°C).

Mean daytime NEE (net ecosystem exchange) of the understory was approximately $0 \text{ g CO}_2 \text{ m}^{-2} \text{ d}^{-1}$ in all sites. The LME model did not indicate a significant effect of site, microform, or the

interaction between site and microform (Table 6). Although there were no statistically significant differences, there were some interesting patterns within the NEE data. At the bog, hollows were consistently a greater source of CO₂, but this effect was muted in the burned treatments (Figure 24). At the fen, hummocks were a greater source of CO₂; however, this effect seemed to shift with burn severity. At the moderately burned fen, NEE had approximately equal values between hummocks and hollows, whereas at the severely burned fen, hollows were a greater source of CO₂, while hummocks appeared to be a net CO₂ sink. The unburned bog had the highest CO₂ ground level emissions from its hollows, while the moderately burned bog, severely burned bog, unburned fen, moderately burned fen, and severely burned fen have similar hollow values. On the other hand, the severely burned fen sequestered the most ground level CO₂ through its hummocks, while all other sites had similar fluxes from hummocks.

Net ecosystem exchange did not have a significant relationship with water table, site, or the interaction between water table and site (Table 6). At the moderately burned bog and unburned fen NEE decreased with a shallower water table, and in all other sites NEE increased with a shallower water table (Figure A 5). There was also no significant relationship between NEE values and soil temperature at 5 cm (Table 6; Figure A 6).

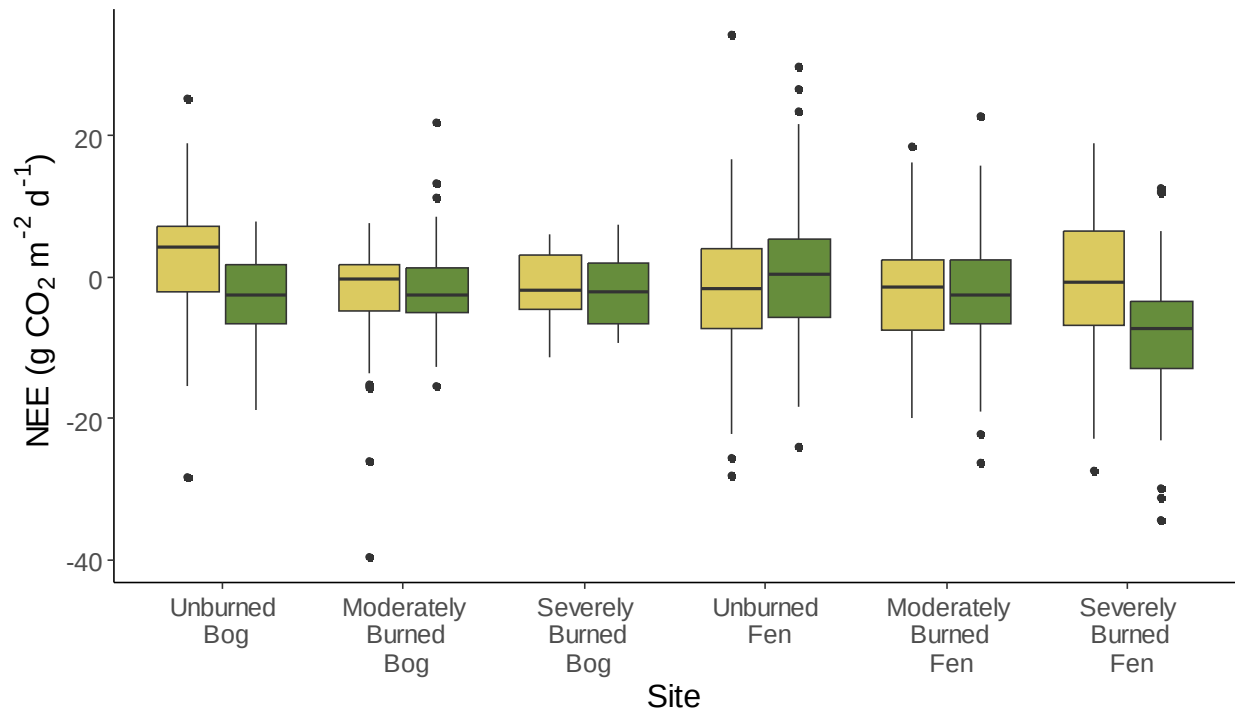


Figure 24: Boxplot depicting the variation in Net Ecosystem Exchange (NEE) across different sites. The y-axis represents NEE while the x-axis shows the different sites sampled. Each boxplot represents the distribution of NEE values within a specific site, with the horizontal line inside the box indicating the median depth, the box encompassing the interquartile range (IQR), and the whiskers extending to 1.5 times the IQR. Outliers, if present, are represented by individual points outside the whiskers.

Overall, CH₄ flux was low from all bog sites and the severely burned fen site. A comparison of both peatlands' CH₄ flux measurements indicated no significant effect of site, microform, or the interaction between site and microform (Table 6), although a pairwise comparison of the same model indicated a significant difference between the severely burned bog and the unburned fen. At the

unburned bog, the hollows tend to consume CH_4 , while the hummocks released approximately equal amounts to what the hollows assimilated (Figure 25). At the moderately burned bog, the same relationship was observed, where hummocks released CH_4 and hollows removed CH_4 , although this pattern was muted compared to the unburned bog. With increasing burn severity across the bog sites, hollows transitioned from sinks to sources, compared to their paired hummocks, which transitioned from sources to sinks. At the moderately burned and unburned fens, the range of flux measurements was greater compared to all other sites. At both sites, hollows release more CH_4 than hummocks; however, at the moderately burned fen the disparity between microforms was amplified compared to the unburned fen. At the severely burned fen, the hummocks released more CH_4 than the hollows, with values comparable to the moderately burned bog.

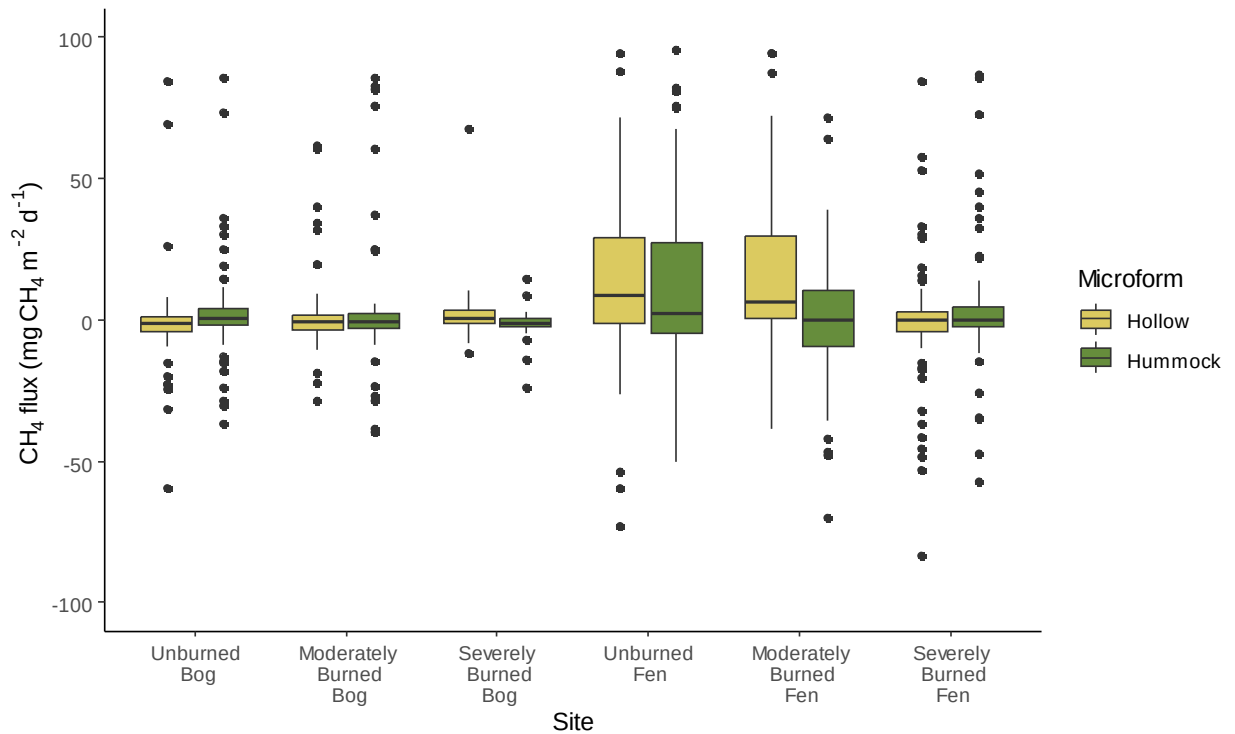


Figure 25: Boxplot depicting CH₄ flux across different sites. The y-axis represents CH₄, while the x-axis shows the different sites sampled. Each boxplot represents the distribution of CH₄ flux values within a specific site, with the horizontal line indicating the median, the box encompassing the interquartile range (IQR), and the whiskers extending to 1.5 times the IQR. Outliers, if present, are represented by individual points outside the whiskers.

An LME assessing the control of environmental variables on CH₄ fluxes indicated no significant effect of water table, site, or the interaction between water table and site (Table 6). At the bog, CH₄ appeared to increase as water table shifted towards and above the surface, with the unburned bog having the most negative slope (Figure 26). Conversely, the moderately burned bog had a slightly positive relationship with water table and CH₄ flux. The severely burned bog once again has a negative

slope, although not as steep as the unburned bog. At the fen, there was a positive relationship between CH₄ flux and water table height, which decreased in slope slightly for the moderately burned fen, and decreased further still for the severely burned fen, which was nearer to neutral.

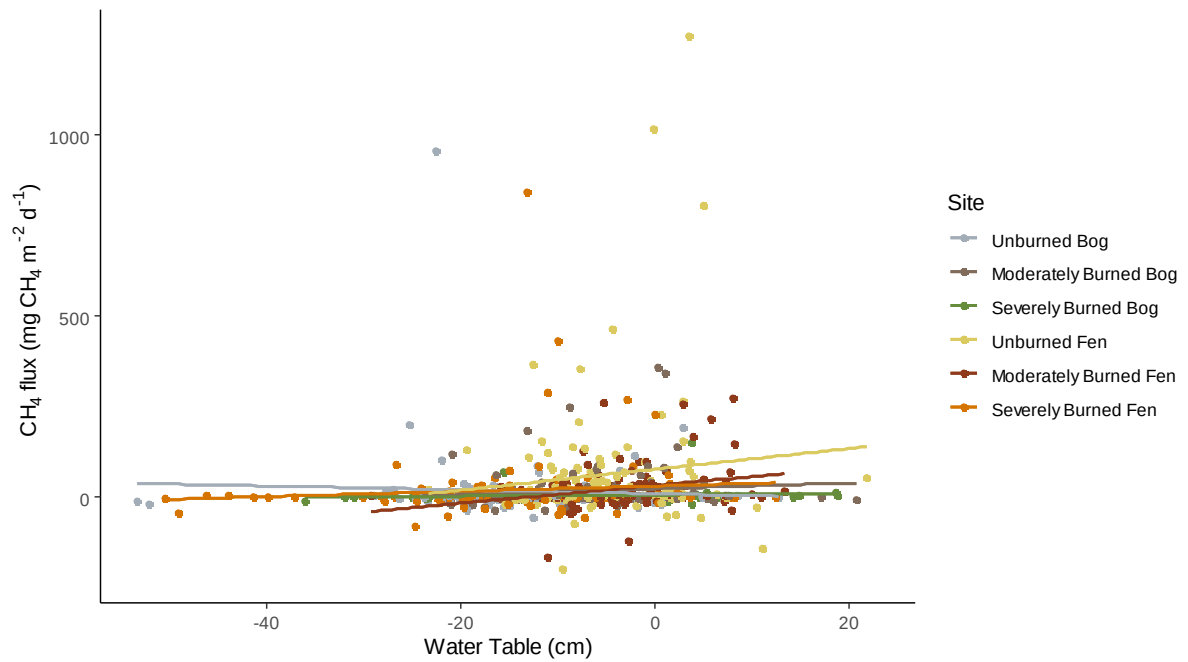


Figure 26: Scatterplot depicting the variation in CH₄ fluxes across different water table depths (cm). The y-axis represents CH₄, while the x-axis represents water table depth (cm).

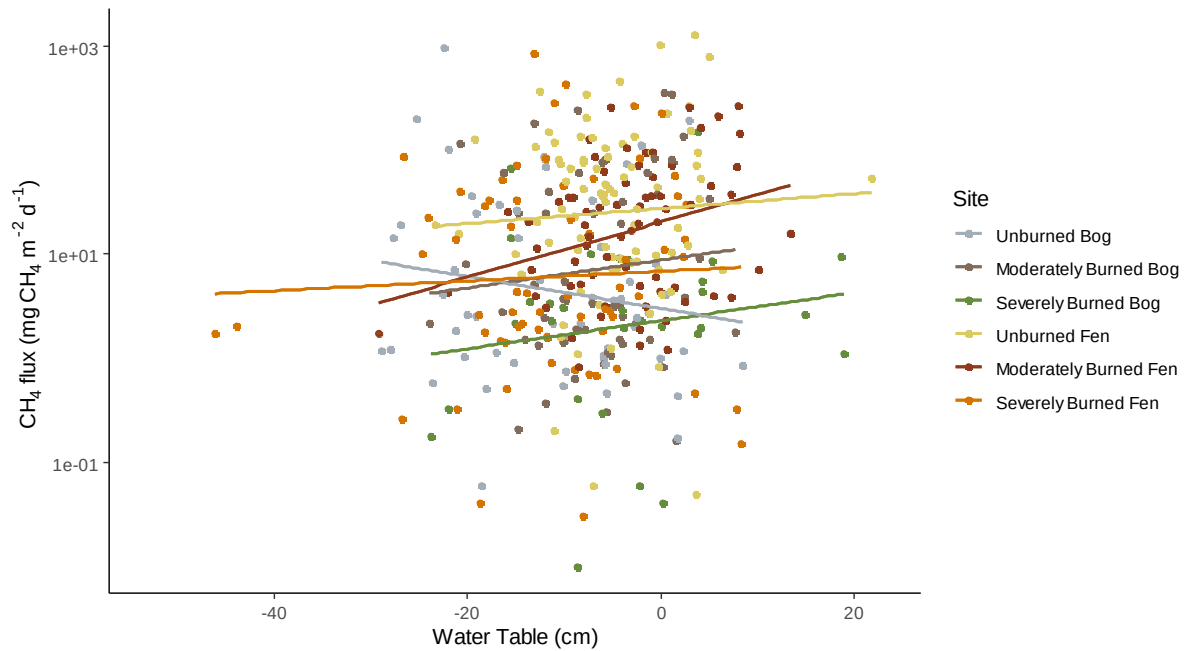


Figure 27: Scatterplot depicting the variation in log scale transformed CH₄ fluxes across different water table depths (cm). The y-axis represents CH₄, while the x-axis represents water table depth (cm).

Along with trends over the growing season (Figure 25), monthly CH₄ trends were also investigated (Figure A 7, Figure A 8), and were deemed non-significant.

An LME model indicated no significant effect of 30 cm soil temperature but did indicate a significant interaction between 30 cm soil temperature and site regarding CH₄ flux (Table 6). At every site other than the severely burned fen there was a positive relationship between CH₄ fluxes and 30 cm soil temperature (Figure 28). At the severely burned fen the opposite was true.

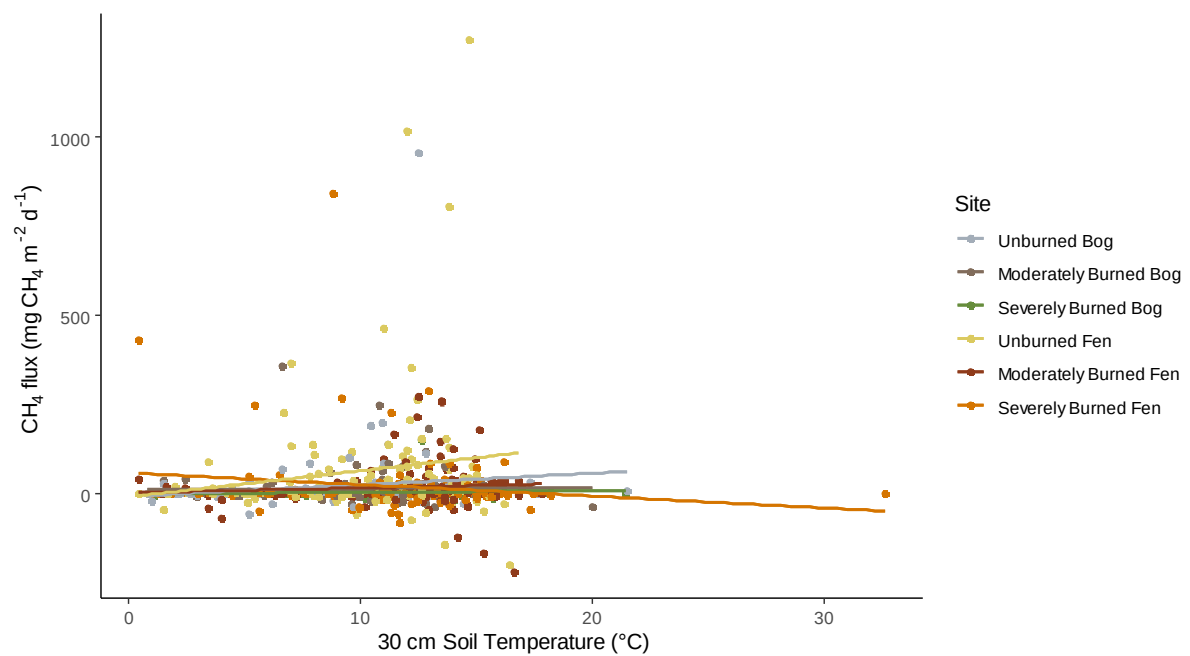


Figure 28: Scatterplot depicting the variation in CH₄ fluxes across different soil temperatures at 30 cm. The y-axis represents CH₄, while the x-axis represents soil temperature (°C).

Table 6: Main effects for CO₂, CH₄, GCC, and environmental variables

	numDF	denDF	F-value	p-value
--	-------	-------	---------	---------

Water Table				
Intercept	1	745	4.947869	0.0264
Microform	1	62	6.110245	0.0162
Site	5	62	0.958278	0.4503
Site * Microform	5	62	0.587796	0.7092
5 cm Soil Temperature	1	714	0.053875	0.8165
30 cm Soil Temperature	1	714	0.700029	0.4031
5 cm Soil Temperature * 30 cm Soil Temperature	1	714	0.822667	0.3647
5 cm Soil Temperature				
Intercept	1	746	276.55846	<.0001
Microform	1	62	11.46379	0.0012
Site	5	62	3.28216	0.0108
Site * Microform	5	62	1.14610	0.3459
30 cm Soil Temperature				
Intercept	1	717	173.91009	<.0001
Microform	1	62	41.24854	<.0001
Site	5	62	15.00858	<.0001
Site * Microform	5	62	5.35086	4e-04
GEP				
Intercept	1	736	30.664224	<.0001
Microform	1	62	0.625873	0.4319
Site	5	62	4.147096	0.0026
Site * Microform	5	62	0.713079	0.6159
ER				
Intercept	1	737	36.26505	<.0001
Microform	1	62	0.24323	0.6236
Site	5	62	6.25490	0.0001
Site * Microform	5	62	0.56133	0.7292
NEE				
Intercept	1	721	0.6594996	0.4170
Microform	1	62	0.0225180	0.8812
Site	5	62	0.6640039	0.6521
Site * Microform	5	62	1.0438095	0.4002
CH₄				
Intercept	1	644	0.8887717	0.3462
Microform	1	36	0.0017897	0.9665
Site	5	36	2.0271918	0.0981
Site * Microform	5	36	0.5464950	0.7398
GEP and Environmental Variables				

Intercept	1	717	8.034994	0.0047
5 cm Soil Temperature	1	717	0.434839	0.5098
Water Table	1	717	0.422147	0.5161
Site	5	68	1.274303	0.2852
5 cm Soil Temperature * Water Table	1	717	0.009864	0.9209
5 cm Soil Temperature * Site	5	717	0.596580	0.7026
Water Table * Site	5	717	0.723533	0.6059
5 cm Soil Temperature * Water Table * Site	5	717	1.624521	0.1511
ER & Environmental Variables				
Intercept	1	1806	16.318438	0.0001
5 cm Soil Temperature	1	1806	0.538318	0.4632
Water Table	1	1806	2.100847	0.1474
Site	5	69	7.186095	<.0001
5 cm Soil Temperature * Water Table	1	1806	3.608993	0.0576
5 cm Soil Temperature * Site	5	1806	4.539356	0.0004
Water Table * Site	5	1806	4.887485	0.0002
5 cm Soil Temperature * Water Table * Site	5	1806	6.589506	<.0001
NEE & Environmental Variables				
Intercept	1	702	0.0091539	0.9238
5 cm Soil Temperature	1	702	0.5376071	0.4637
Water Table	1	702	0.0305054	0.8614
Site	5	68	1.5219985	0.1946
5 cm Soil Temperature * Water Table	1	702	0.0148656	0.9030
5 cm Soil Temperature * Site	5	702	1.2793022	0.2709
Water Table * Site	5	702	0.4939727	0.7809
5 cm Soil Temperature * Site * Water Table	5	702	0.7082749	0.6173
CH₄ & Environmental Variables				
Intercept	1	550	0.0128895	0.9097
30 cm Soil Temperature	1	550	0.1732477	0.6774
Site	7	52	0.8309180	0.5665
Water Table	1	550	0.0424520	0.8368
30 cm Soil Temperature * Site	7	550	2.6181799	0.0115
Water Table * 30 cm Soil Temperature	1	550	0.0013679	0.9705
Water Table * Site	7	550	0.6984338	0.6734
30 cm Soil Temperature * Water Table * Site	7	550	1.6064595	0.1309
GCC				
Intercept	1	923	2179.5349	<.0001
Microform	1	36	0.0086	0.9268
Site	5	36	8.0254	<.0001

Site * Microform	5	36	1.0345	0.4124
------------------	---	----	--------	--------

Table 7: Main effects of collar vegetation surveys

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Collar Vegetation Percent Cover					
Site	5	522	104.3	0.1334	0.9846
Microform	1	187	187.36	0.2395	0.6250
Site * Microform	5	970	193.91	0.2478	0.9407

3.4 Discussion

3.4.1 CO₂

The objectives of this study were to quantify and compare surface-level C fluxes between two peatland types, a bog and a fen, along a fire severity gradient. There were two main hypotheses (1) seven years following a wildfire, moderately burned sites would be further along the recovery trajectory than the severely burned sites, meaning they would be more similar to the unburned treatments in terms of ground level C fluxes, and (2) that the bog would be further along the recovery trajectory than the fen in terms of C fluxes. This was first assessed by observing patterns of gross ecosystem productivity (GEP) within and between sites. The former hypothesis was informed by the straightforward belief that the intensity of the disturbance would be directly associated with the timeline for recovery. In contrast, the latter hypothesis was guided by previous research, which suggested that the richer nutrient status of fens could allow invasive or opportunistic vegetation to outcompete pre-fire vegetation communities (Grau-Andrés et al. 2019), and that the greater microtopographic variability of bogs (Sumner 2007) could confer greater resilience to these peatland ecosystems.

Gross ecosystem productivity was significantly greater at Poplar Fen than Prairie Creek Bog across all sites and burn treatments. When the green chromatic coordinate (GCC) was compared

between sites, there was a higher GCC at Poplar Fen sites regardless of burn severity, suggesting a higher amount of chlorophyll found at the surface level capable of photosynthesizing, and sequestering C, which is true of undisturbed fens in most instances (Zoltai and Vitt 1995). Fens are commonly found to be much more productive than bogs pre-disturbance, with average GEP rates of 5.1-6.9 g C m⁻² day⁻¹, while bogs have average GEP rates of 3.8-4.3 g C m⁻² day⁻¹ (Sulman et al. 2010). Following wildfire disturbance, fens have been observed to be notably less productive, with lower rates of GEP 1-year post-fire (Morison et al. 2021). Interestingly, Morison et al. (2021) saw an increase in CO₂ uptake along the fire severity gradient, with deeper burns sequestering more CO₂ than comparatively shallower burns, which is consistent with the Lukenbach et al (2016) conceptual model. It has been speculated that fens are expected to recover more quickly than bogs (Davidson et al. 2019), but as of yet there are no empirical studies on the topic. Bogs typically become sources of C, returning to sinks after about 10-13 years post-fire, as the moss, shrub, and tree species recover (Wieder et al. 2009). Comparatively, in the present study rates of productivity increased significantly at the fen and decreased at the bog, suggesting that bogs may be slower to recover than previously thought. This can largely be attributed to divergent changes in the plant community following wildfire, which behave differently depending on wetland class and fire severity (Chapter 2; Benscoter 2006; Lukenbach et al. 2016).

Plant communities seem to follow different trajectories in response to wildfire. Post wildfire, fens undergo a period of abundant vegetation growth (Chapter 2), greatly increasing GEP along the increasing fire severity gradient. Transitions in the plant community toward shrubs and tree seedlings encourage rapid accumulation of ground level C. In the severely burned fen, there was an increase in forb and shrub percent coverage (Figure 17), which ground level C fluxes are capable of capturing. It is possible that plant size allowed for a more representative account of the entire plant community at

the severely burned fen, as chamber size allowed for a maximum plant height of ~40 cm. At the unburned fen and moderately burned fen, there are more tall shrubs and established trees which hindered the possibility of truly representative site-scale C flux measurements, only allowing for ground level C fluxes.

On average, GEP was statistically the same at the unburned bog, moderately burned bog, and severely burned bog, meaning that statistically, each ecosystem is equally as productive as the next at sequestering C at the ground level. Although productivity is the same across each bog treatment, the unburned bog has a positive, though non-significant, relationship between GEP and water table depth, which is expected. As the water table rises closer to the moss surface, peatland plants have easier access to water, which is especially beneficial for non-vascular plants that rely on capillary action to transport water and are the dominant plant type of bogs (Zoltai and Vitt 1995). Unusually, the moderately and severely burned bogs both have a negative relationship between GEP and water table depth. This may indicate a reduction in water storage capacity because of a loss of surface peat, and the ability of species assemblages in unburned peatlands to moderate the water table depending on the conditions of the environment around them, which may be affected by burning (Lukenbach et al. 2016; Elmes et al. 2019). This relationship relates to GCC, as greenness indices were higher at the unburned bog than the moderately burned and severely burned bog sites (Figure A 3; Figure A 4).

The specific yield of shallow near-surface peat is much higher than deeper in the peat profile, which has experienced more advanced decomposition and been compressed by overlying organic material. This leaves fewer, smaller pore spaces for water, resulting in even minor additions of water to cause dramatic changes to the water table (Thompson and Waddington 2013). On the other hand, the portion of peat closer to the surface has a higher specific yield, generally exhibiting larger pores that

drain at lower tensions and thus have a large capacity to store water and ameliorate the response of the water table to individual precipitation events (Thompson and Waddington 2013). The disturbance in the moderately and severely burned bog sites burned the upper portion of peat, creating an environment where the surface layer of peat has a much lower specific yield compared to unburned peatlands. This change to the environment may lead to more dramatic fluctuations in the water table, altering the expected relationship between GEP and water table, which is an increase in GEP flux measurements with an increase in water table. The water table at the burned sites may exhibit large fluctuations with precipitation events, inundating the site, preventing productivity due to flooding, which would quickly return to the lowered water table position upon cessation of rainfall. Under these disturbed conditions, the degraded peat properties could lead to far lower water table positions during periods of drought, which were the climatic conditions in the two years studied.

At Poplar Fen, the rates of GEP between the unburned and moderately burned treatments were not statistically different, but the severely burned fen was significantly more productive than any other site. In the fen, a higher water table leads to lower productivity at the unburned site, and greater productivity at the moderately and severely burned sites. In this case, the burned treatments were behaving as expected, increasing productivity with easier access to the water table, while the unburned fen is responding unusually to changes in the water table. These relationships may also be related to differences in the optimal water table depth at each site. Each site may have different water needs depending on community compositions, species abundance, and time since disturbance. In some cases, older growth may prefer a lower water table that, for instance, developed trees are able to access while avoiding rotting. On the other hand, a younger growth site might prefer a water table closer to the surface, where shallower rooting vascular vegetation can access water, and non-vascular vegetation can

draw water up through capillary action. Thus, differences in GEP may relate to each sites optimal water table depth, and while informed hypothesis can be made based on averages of optimal water depths from sites with similar plant communities, it is difficult to compare burned sites to unburned sites without a more thorough understanding of the soil moisture dynamics of each site. In regards to the fen, perhaps the water table exists above the optimal depth for the unburned fen throughout the growing season, causing it to respond differently to increases or decreases in the water table than the burned sites, which align more closely with the expected behavior of an undisturbed peatland. A longer-term monitoring record with a greater variety of meteorological and thus water table conditions could help elucidate this relationship.

Ecosystem respiration (ER), like GEP, was not significantly different across the burn severity gradient at the bog or the fen. Ecosystem respiration was, however, significantly greater at the fen compared to the bog, tending to increase with the fire severity gradient, while at the bog ER tended to decrease as burn severity increased. A lower ER is commonly noted in bogs, with fluxes in the range of $3.3\text{--}3.7 \text{ g C m}^{-2} \text{ day}^{-1}$, while fen ER rates have been observed in the range of $4.2\text{--}4.8 \text{ g C m}^{-2} \text{ day}^{-1}$ (Sulman et al. 2010). As the fen sites gain woody biomass, GEP and ER increase, however, since the bog is slow to recolonize, GEP and ER remain low in comparison. Much like the relationship with GEP, ER follows the same trends with GCC, which indicates a higher rate of organic matter deposition as well as productivity. Although this assumes a higher greenness index indicates greater seasonal surface level plant growth, which would lead to greater organic matter deposition in the fall. Trends in ER lead to the same findings as GEP, that plant respiration is a crucial indicator of ecosystem recovery trajectories.

At the bog, the relationship between water table depth and ER was positive across the burn severity gradient, although it is more similar between the unburned and moderately burned treatments. This is an unexpected relationship. Throughout the literature it is stated that as the water table depth increases, the aerated portion above the water table will exhibit a commensurate increase, thus so too does aerobic respiration (Schlesinger and Andrews 2000; Dinsmore et al. 2009; Amiro et al. 2010; Kechavarzi et al. 2010; Treat et al. 2014; Jassey et al. 2018 Mar; Laine et al. 2018; Susilawati et al. 2022). At the fen, the unburned and severely burned treatments both exhibit a more conventional negative relationship between ER and water table, where a water table closer to the soil surface is related to reduced rates of ER. The moderately burned fen differs from the unburned and severely burned treatments and has a positive relationship between ER and water table, much like the bog sites. Although not significantly different, ER values are the greatest at the severely burned fen. The severely burned fen also had significantly higher 5 cm soil temperatures compared to the other five sites, which may contribute to faster rates of decomposition above the water table, as increased temperatures are known to lead to increased decomposition rates (Susilawati et al. 2022). The higher near-surface soil temperature at the fen is likely due to a decrease in albedo.

Unburned peat can access and draw water to the surface from a water table level of approximately 1 m (Thompson and Waddington 2013). Any lower than this and the capillary forces are no longer strong enough to draw water to the surface. This threshold changes in burned peat, with a higher bulk density, and lower volume of water loss or gain required to make dramatic changes in the water table position. It is possible that during the two seasons in which measurements took place, which were dry years compared to the 30-year average, that the water table often existed at these sites below the 1 m threshold, and that precipitation events allowed typical functioning of the peatland, including

ER, to proceed. In this way, it would appear as though respiration increases in flooded conditions, however, if the bog were in a water deficit, water inputs may allow plant and soil respiration to proceed. This is supported by the classification of bogs as precipitation-fed peatlands as, with no other water input, the patterns and quantity of rainfall has a considerable influence on ecosystem function. Since Prairie Creek Bog was located roughly 60 km south of Poplar Fen, it is quite likely that they were impacted by different individual precipitation events, while still experiencing comparable long-term climates. At Poplar Fen, the moderately burned site also had a positive relationship between ER and water table, differing from the other two fen sites. At the moderately burned fen, the water table had the lowest range of measured water table values compared to all other sites, perhaps the effect of only a few precipitation events impacted these values.

As mentioned with the bog, it is possible that there was a water deficit at the moderately burned fen affecting ER flux measurements. However, this would more likely impact all three fen sites, since they exist within the same peatland complex. An important source of variation between fens across the burn severity gradient is vegetation community composition. The unburned and severely burned fen sites have a much more similar composition, while the moderately burned fen has nearly twice as much above-ground biomass, and a far greater proportion of shrubby species. It has been shown that *Betula glandulosa*, which is the dominant shrub species at the moderately burned fen, contributes significant amounts of labile C to soil, contrasting with other shrub species which allocate photosynthate to belowground biomass (Street et al. 2018). Thus, this addition of C may have contributed to the soil throughout the growing season, but perhaps contributes to greater ER when the water table rises.

Net ecosystem exchange (NEE) was not statistically different across any of the six sites, consistent with the findings of Morison et al. (2021). Since both GEP and ER were greater at Poplar

Fen than at Prairie Creek Bog, NEE was approximately zero at all six sites. Water table and 5 cm temperature, although not significant, have an interesting relationship with NEE. Across all six sites, only the unburned fen and severely burned bog show an increase in NEE with an increase in 5 cm soil temperature. Conversely, only the moderately burned bog and unburned fen have decreasing NEE flux values with increasing water table. Compared to the relationships of the two environmental variables to GEP, the unburned fen, moderately burned bog, and severely burned bog all have lower rates of productivity with a higher water table, supporting the assertion that the high water table observed during the study period may be outside of the optimal water table depth for these sites.

Net ecosystem exchange measurements in this study importantly only depict understory fluxes due to constraints on chamber size. Differences in NEE six to seven years after fire are not major, nonetheless there is clear evidence of rapid recovery in the fen ecosystem. Overstory productivity is not included in this assessment, yet it plays an important role in recovery of C uptake in peatlands post-wildfire (Wieder et al. 2009). Tree surveys at each of the study sites (Chapter 2) indicate that tree and tall shrub biomass at the moderately and severely burned fens is equal to or greater than the unburned fen, providing further evidence of rapid recovery of C uptake in the overstory. In contrast, overstory tree biomass at the burned bog sites remained much lower than the unburned bog, indicating slower recovery. Nonetheless, tree cover in the ground layer was highest at the severely burned bog and similar at moderately burned and unburned bogs. This indicates tree seedling establishment at six to seven years post-fire that will eventually return overstory C uptake. This is consistent with *P. mariana* recovery in a chronosequence of bogs in northern Alberta in which tree biomass remained low for several decades post-fire (Wieder et al. 2009).

3.4.2 CH₄

CH₄ emissions were low at all three Prairie Creek Bog sites, as well as the severely burned fen, but more than double at the unburned and moderately burned fen sites. These values are generally consistent with the literature, as bogs are typically drier, and thus have lower CH₄ fluxes than fens (Turetsky et al. 2014). At the unburned and moderately burned fens, there are also markedly more graminoid species present (Figure 17), although this provides little explanation into the reason for such little CH₄ production at the severely burned fen, which also has some graminoid growth. Currently, there is little data available on CH₄ fluxes from peatlands following wildfire; however, Davidson et al. (2019) found that, particularly after severe burning, emissions of CH₄ from a fen were far lower compared to an unburned control.

At each site other than the severely burned fen, there is a positive relationship between CH₄ flux and 30 cm soil temperature. Soil temperature has been shown to be positively related to CH₄ efflux (van Hulzen et al. 1999) by increasing the rate of organic matter degradation (Segers 1998). However the microbial community itself is responsive to changes in the water table, which facilitate or inhibit growth of the community and a hospitable environment (Bridgham et al. 2013). At the severely burned fen, the negative relationship between CH₄ flux and soil temperature may be a result of the soil temperature-water table-relationship, whereby when water table depth increases, soil temperature decreases. The rise in water table has a deleterious effect on the influence of soil temperature, leading to a negative relationship between CH₄ flux and water table.

CH₄ is strongly associated with vegetation, as one method of CH₄ emission is plant mediated transport. In regard to the fen, the severely burned site had the greatest amount of C stored in

aboveground tree biomass (Chapter 2) and had the lowest graminoid percent cover (Figure 17). This may explain why the severely burned fens' fluxes are so vastly different from the unburned and moderately burned fens. It is possible that due to a greater graminoid cover, plant mediated transport of CH₄ (Dinsmore et al. 2009) is much higher at the unburned and moderately burned fen treatments. In comparison to the bog, there are few to no graminoids present. The unburned and moderately burned fens also have approximately one sixth of the tree biomass present at the severely burned fen, which may allow new graminoid growth and propagation. Previous studies have noted the ability of graminoids to transport CH₄ through aerenchymatous tissue (Bhullar et al. 2013), suggesting that this additional pathway for CH₄ to enter the atmosphere may be responsible for increased CH₄ efflux.

It is also possible that the CH₄ trends in the bog and fen are signs of recovery. In 2017 and 2018, Davidson et al. (2019) measured CH₄ at Poplar Fen and came to the conclusion that wildfire had overridden the hydrological controls that normally are related to CH₄ cycling, and that despite differences in the relationship between fluxes and the water table, an increase in CH₄ emissions was indicative of the moderately burned fen beginning to behave more like the unburned fen. This can also be seen at the bog, albeit in much lower concentrations, where the hummocks and hollows at the moderately burned bog are behaving more like the hummocks and hollows of the unburned bog. Davidson et al. (2019) suspected that the severe burn had likely removed a high proportion of the soil microbial community responsible for both producing and oxidizing CH₄, while the moderate burn reduced the microbial load, but not completely, so that they were able to return more quickly than at the severely burned site. This is probable, although cannot be confirmed in this study as no samples of the microbial community were collected.

3.5 Conclusion

Ground layer C fluxes and vegetation were compared at both a bog and a fen along a burn severity gradient. The outcome of these measurements revealed interesting relationships in CO₂ and CH₄ along the fire severity gradient. At the fen, GEP was significantly higher than at the bog, which correlated directly with plant community greenness, measured as GCC (Figure 16, Figure A 3, Figure A 4). Higher ER at the fen can also be attributed to higher GCC at the sites indicating the importance of plant respiration. GCC as well as plant vegetation surveys suggest that fen ecosystems may be able to recover from wildfire expeditiously compared to bog ecosystems, as recovery is accelerated by the timely growth of aboveground biomass, particularly, fast growing shrubs. The vegetation community, though much different from the unburned control, allows CO₂ to be sequestered in amounts akin to the unburned control. The bog, on the other hand, is apparently recovering more slowly from fire, necessitating the growth of the understory and overstory to return to pre-burned C cycling conditions. Recovery in terms of CO₂ cycling returning to rates comparable to pre-burn conditions is highly dependent on vegetation growth and biomass accumulation.

Although CO₂ cycling appears to have been restored at Poplar Fen rather than at Prairie Creek Bog, CH₄ cycling is contingent not on peatland type, but on burn severity. When burned moderately, recovery of CH₄ cycling to patterns similar to those prior to burning, as inferred from the control site, appears to occur within 7 years. When burned severely, it appears that the soil microbial community and organic matter quality changes slowly. Thus, the recovery of CH₄ cycling to pre-fire conditions depends on other successional strategies.

Chapter 4

Conclusion

The main objectives of this study were to compare the recovery trajectories of bogs and fens six to seven years following wildfire, where recovery refers to a return to pre-burn plant community traits and composition, and functional return of surface-level C fluxes. The first (1) hypothesis posited that six to seven years following wildfire, the moderately burned treatments would exhibit greater recovery than the severely burned treatments. This hypothesis was informed by the assumption that there is a direct relationship between disturbance and time, where a lower severity disturbance takes less time to recover, and a more severe disturbance takes longer. Hypothesis (2) was that the bog would recover faster than the fen. Previous research suggests that invasive or opportunistic vegetation might outcompete pre-fire vegetation communities as a result of the richer nutrient status of fens (Grau-Andrés et al. 2019), and that the greater microtopographic variability of bogs (Sumner 2007) could confer greater resilience to these peatland ecosystems. Over the sampling period, vegetation surveys were conducted, biomass was collected, plant functional traits were measured, and CO₂ and CH₄ fluxes were observed, with the purpose of comparing these variables between a bog and a fen, six to seven years following wildfire, along a fire severity gradient. Here, I summarize key findings, and discuss the implications for future research and modelling efforts.

4.1 Plant community composition, biomass, and plant functional traits

Vegetation grew in far greater abundance at the fen than at the bog. Based solely on the post-fire vegetation development, this study would suggest that the moderately burned fen seemed well on

its way to recovery, as biomass accumulation appeared to be similar to that of the unburned fen. The word recovery, however, is nuanced and is highly dependent on what the desired outcome of recovery is. If recovery means the return of above-ground biomass, then the data would indicate that the moderately burned fen has recovered. However, recovery was contextualized as a return of both abundance and composition of the pre-burned plant community, to which the moderately burned fen has not yet fulfilled. The severely burned fen did not have the same rapidly colonizing species as the moderately burned fen, which may be due to the impact of fire on the seedbank and microbial community that was assumed to have been reduced with burning, and which seems to have not yet returned to pre-burn amounts. The severely burned fen is more like the unburned fen in terms of community composition; however, the abundance of plant matter is greatly reduced. Comparing the community composition of each site, forb abundance was lowest in the unburned fen and increased with increasing burn severity, while forb abundance was high in the unburned bog, decreasing to the severely burned bog. This is contrary to the findings of Grau-Andrés et al. (2019), who observed an increase in forb presence along a fire severity gradient in a raised bog. These findings also diverged from Shepherd et al. (2021), who found no significant interactions between burn severity and habitat type, suggesting uniform wildfire impacts on species across different land classifications. Our study indicates that species recovery varies among ecosystem types, with distinct recovery trajectories for each peatland class. Moss cover was observed to be highest in unburned areas, declining as burn severity increased, culminating in the lowest cover at severely burned sites. The unburned bog had nearly twice as much moss than the moderately and severely burned bogs. Similarly, shrub cover was most prevalent in unburned bog, decreasing progressively towards severely burned bog. At the fen, shrub percent cover was highest at the moderately burned treatment. Leaf litter had the greatest accumulation in the severely burned fen, diminishing towards the unburned fen. Tree seedling cover,

while not statistically significant, tended to be greater in severely burned areas compared to moderately burned ones.

Biodiversity indices revealed further nuances in community structure. Shannon's diversity index ranked the sites from highest to lowest diversity as follows: severely burned bog, moderately burned bog, unburned bog, unburned fen, moderately burned fen, and severely burned fen. Similarly, Simpson's diversity index ranked the sites: severely burned bog, moderately burned bog, unburned fen, unburned bog, moderately burned fen, and severely burned fen. Thus, wildfire appears to result in higher diversity at the bog but reduces diversity at the fen.

Furthermore, plant functional traits differed between sites in several ways. *P. mariana* seedlings tended to grow taller at the moderately burned fen than they did at the severely burned fen, unburned fen, and unburned bog, although with a lower LDMC, whereas *P. mariana* growing at the unburned bog and fen had high LDMC, indicating high resource availability. At the unburned sites, *P. mariana* distributes resources to moderate heights and SLA in comparison to the burned sites. At the moderately burned bog, *P. mariana* seedlings had a similar strategy, with moderate upward growth and increasing SLA. At the moderately burned fen, *P. mariana* expended resources on rapid elongation, becoming the tallest on average across all five sites. Although resources may be limited at the severely burned fen, *P. mariana* allocated more resources to SLA than height, growing shorter, but with greater SLA than others along the burn severity gradient. Although non-significant, *R. groenlandicum* exhibited similar strategies to *P. mariana* in the unburned treatments, allocating resources to grow to average heights, with modestly-sized leaves compared across sites. LDMC was low at the moderately burned fen, but in similar manner to *P. mariana*, resources were predominantly used for rapid expansion, although outward rather than upward, thus filling a different spatial niche. *B. glandulosa* is

approximately twice as tall at the moderately burned fen than the unburned and severely burned fen sites and allocated fewer resources to leaf growth with increasing burn severity, as resources grow more limited as indicated by a lower LDMC.

In the bog, vegetation recovery was slower, but the severely burned treatments showed promising signs of tree seedling presence, suggesting a possible recovery over decadal timescales (Wieder et al. 2009). While both the moderately burned bog and fen appeared to be on a path to recovery in terms of C fluxes, the vegetation communities of the severely burned sites were more akin to the unburned conditions. This dichotomy between vegetation and C flux data indicates that the severely burned sites may be closer to full recovery in terms of vegetation structure, as the communities are more similar to the unburned treatments, while C dynamics suggest the opposite, since C fluxes are more similar between the moderately burned treatments and the unburned treatments. Regarding the second hypothesis, the fen seems closer to recovery than the bog.

The plant functional traits were measured as potential proxies for recovery. The findings suggest that while these traits provide valuable insights, they may not fully capture the complexities of post-fire recovery dynamics. There are challenges associated with using plant functional traits as a proxy at this point this is one of few studies that consider plant functional traits, so data on what can be expected are minimal. This underscores the need for extensive data collection of plant functional traits spanning several years at undisturbed sites, to understand what the range of typical pre-disturbance conditions, so that post-fire changes may be interpreted more accurately. Overall, the data from moderately burned bogs suggest that the sites may be progressing toward recovery at the six-to-seven-year mark, though severely burned areas may require longer. These findings align with the assertion of Wieder et al. (2009)'s assertion that bogs typically recover within 10-13 years.

4.2 Carbon cycling

Greenness was significantly greater at the fen than at the bog, which was related to the measured greater quantity of biomass, as greater biomass would suggest more chlorophyll, and thus high greenness indices. This was likely the driver of the greater productivity, with the fen having significantly lower (negative values indicating CO₂ uptake) GEP values than the bog. The interaction between water table and site for explaining variation in GEP was significant. At the moderately burned and severely burned bog sites, as well as the unburned fen, GEP decreased with an increasing water table (closer to the moss surface). The opposite was true of the severely burned fen, moderately burned fen, and unburned bog, where GEP decreases with increasing water table.

ER was significantly greater at all fen sites compared to all bog sites. Variation in ER was significantly explained by 5 cm soil temperature and site, and water table and site interactions. All three bog treatments, as well as the moderately burned fen had a positive relationship between water table and site. The unburned and severely burned fens, however, have a negative relationship between water table and ER, where an increase in water table (closer to the moss surface) was correlated with a decrease in ER (greater values indicate greater release of CO₂). At the moderately burned fen, the increase in 5 cm soil temperature was related to a decrease in ER.

Overall, CH₄ flux was low from all bog sites and the severely burned fen site. At the unburned bog, the hollows tended to consume CH₄, while the hummocks released roughly equal amounts to what the hollows sequestered (Figure 25). At the moderately burned bog, the same relationship was observed, with hummocks releasing CH₄ and hollows consuming CH₄, although this pattern was muted compared to the unburned bog. Across the bog sites, with burn severity increasing, hollows transitioned from sinks to sources, compared to their paired hummocks, which transitioned from sources to sinks. At the

moderately burned fen and unburned fen, the range of flux measurements was greater compared to all other sites. At both sites, hollows released more CH₄ than hummocks; however, at the moderately burned fen the disparity between microforms was amplified compared to the unburned fen. At the severely burned fen, the hummocks released more CH₄ than the hollows, with values comparable to the moderately burned bog.

This study's results partially supported the hypotheses. Matching expected patterns, the fen exhibited greater total biomass six to seven years post-fire, which initially suggested a more advanced recovery trajectory. However, "recovery" entails more than just biomass and C accumulation. The moderately burned fen, although possessing more biomass, showed a significant shift in vegetation composition compared to the unburned fen, potentially indicating a new stable state rather than a return to pre-fire conditions, or at least is suggestive of a longer successional trajectory. In contrast, the severely burned fen, with its sparse vegetation, more closely resembled the unburned fen in composition but lagged in biomass recovery. The limited availability of CH₄ measurements for comparison across peatlands further complicates our understanding of these dynamics. This gap in data emphasizes the necessity for additional research to provide a clearer picture of CH₄ flux variability and its driving factors post-fire.

The results and interpretations of this study must be considered within the context of error and uncertainty, which can propagate through the work at each stage, from conceptualization, methodology, data collection, and analysis. These measurements are averaged over only two years, and two dry years at that. As pre-fire conditions are not available across all treatments, it is not possible to determine if the source of the variation in C fluxes exists because of fire alone or also due to inherent differences among the study sites. Further, longer records of C fluxes are necessary to dampen the confounding

influence that interannual differences in climate have. Nevertheless, differences in the response of C fluxes to environmental conditions among sites suggest that fire severities contributed significantly to the patterns observed. Additionally, ground-level C fluxes capture only a small portion gas fluxes from the entire ecosystem. While eddy covariance methods would have allowed for measurement at ecosystem scale, the patchiness of fire severity across the landscape makes it difficult to find large enough areas with homogeneous conditions to allow for comparison of fire severities. This limitation was accounted for with tree inventory surveys to provide a proxy of total C held in tree biomass; however, seasonal fluxes can not be obtained from these measurements, and so the flux data cannot be readily compared, as the units of measurement are vastly different. Also, the study would have benefited from soil moisture and soil water tension measurements. While the water table is a reasonable proxy for near-surface soil moisture dynamics, it is predicated upon assumptions that can often not be true at any given point in time and that may change in response to fire-induced changes in soil hydrophysical properties. Finally, microbial community assessments would have removed the need to speculate about their presence or absence/activity level. Full genetic surveys may not have been necessary, but measurements of the total microbial community biomass would have provided certainty around assumptions that have been made throughout the study.

4.3 Implications of the results for modelling post-fire carbon cycling in peatlands

These findings underscore the need for more nuanced consideration of burn severity in peatland management and research. Future studies should include chronosequences in fen ecosystems to better estimate recovery timelines, similar to the study Wieder et al. (2009) performed with bogs. Severe burns are becoming more common with climate change and results from this study suggest that post-fire

recovery varies depending on burn severity. There is an immediate need to further our understanding of recovery in these systems, so that burn severity may be implemented into global C models. This study also highlights the importance of distinguishing between bogs and fens in ecological modeling, as applying the same rates of C efflux or accumulation could lead to significant inaccuracies. Understanding these differences is crucial for accurate modeling and prediction of C dynamics in peatland ecosystems, particularly in the context of increasing wildfire frequency and intensity with climate change.

4.4 Future research needs

There are several avenues for future research that have been identified by this study. First, long-term data is needed for plant functional traits and CH₄ flux data from undisturbed peatlands, so that there is a better idea of what the fluxes and plant functional traits looked like pre-disturbance. Data is needed at more sites and over longer time periods to minimize the confounding effect of interannual climatic variability. Finally, more data on plant communities, hydrology, and C cycling is needed in fen ecosystems affected by wildfire. This study has illustrated the large differences in flux data between peatland classes, and the necessity to distinguish between peatland types when considering response to disturbance. As northern fire regimes are shifting, becoming more frequent and intense, it is imperative that future studies consider more severe fires, or burns along a severity gradient, as there are clear differences in recovery.

The cycle of wildfire disturbance and recovery that has defined Canada's boreal region for millennia can not necessarily be depended upon in the future. A different climate not only brings more frequent and severe wildfire but also alters the conditions that recovery happens under. The compounded effect of more intense wildfire and altered climate could steer the recovery trajectory away

from pre-disturbance conditions. Six to seven years post-fire, the severe burns in this study remain functionally unrecovered. We need to better understand peatland ecosystem recovery and the characteristics that influence it now more than ever. This study advances this understanding.

References

- Alm J, Talanov A, Saarnio S, Silvola J, Ikkonen E, Aaltonen H, Nykänen H, Martikainen PJ. 1997. Reconstruction of the carbon balance for microsites in a boreal oligotrophic pine fen, Finland. *Oecologia*. 110(3):423–431. doi:10.1007/s004420050177.
- Amiro BD, Barr AG, Barr JG, Black TA, Bracho R, Brown M, Chen J, Clark KL, Davis KJ, Desai AR, et al. 2010. Ecosystem carbon dioxide fluxes after disturbance in forests of North America. *J Geophys Res Biogeosciences*. 115(G4):2010JG001390. doi:10.1029/2010JG001390.
- Armstrong A, Waldron S, Ostle NJ, Richardson H, Whitaker J. 2015. Biotic and Abiotic Factors Interact to Regulate Northern Peatland Carbon Cycling. *Ecosystems*. 18(8):1395–1409. doi:10.1007/s10021-015-9907-4.
- Ballhorn U, Siegert F, Mason M, Limin S. 2009. Derivation of burn scar depths and estimation of carbon emissions with LIDAR in Indonesian peatlands. *Proc Natl Acad Sci*. 106(50):21213–21218. doi:10.1073/pnas.0906457106.
- Benscoter BW. 2006. Post-fire bryophyte establishment in a continental bog. *J Veg Sci*. 17(5):647–652. doi:10.1111/j.1654-1103.2006.tb02488.x.
- Benscoter BW, Vitt DH. 2008. Spatial Patterns and Temporal Trajectories of the Bog Ground Layer Along a Post-Fire Chronosequence. *Ecosystems*. 11(7):1054–1064. doi:10.1007/s10021-008-9178-4.
- Benscoter BW, Wieder RK. 2003. Variability in organic matter lost by combustion in a boreal bog during the 2001 Chisholm fire. *Can J For Res*. 33(12):2509–2513. doi:10.1139/x03-162.
- Bhullar GS, Irvani M, Edwards PJ, Olde Venterink H. 2013. Methane transport and emissions from soil as affected by water table and vascular plants. *BMC Ecol*. 13(1):32. doi:10.1186/1472-6785-13-32.
- Bona KA, Shaw C, Thompson DK, Hararuk O, Webster K, Zhang G, Voicu M, Kurz WA. 2020. The Canadian model for peatlands (CaMP): A peatland carbon model for national greenhouse gas reporting. *Ecol Model*. 431:109164. doi:10.1016/j.ecolmodel.2020.109164.
- Bowman DMJS, Balch JK, Artaxo P, Bond WJ, Carlson JM, Cochrane MA, D'Antonio CM, DeFries RS, Doyle JC, Harrison SP, et al. 2009. Fire in the Earth System. *Science*. 324(5926):481–484. doi:10.1126/science.1163886.
- Bridgman SD, Cadillo-Quiroz H, Keller JK, Zhuang Q. 2013. Methane emissions from wetlands: biogeochemical, microbial, and modeling perspectives from local to global scales. *Glob Change Biol*. 19(5):1325–1346. doi:10.1111/gcb.12131.
- Brown CD, Johnstone JF. 2012. Once burned, twice shy: Repeat fires reduce seed availability and alter substrate constraints on *Picea mariana* regeneration. *For Ecol Manag*. 266:34–41. doi:10.1016/j.foreco.2011.11.006.
- Brown LE, Palmer SM, Johnston K, Holden J. 2015. Vegetation management with fire modifies peatland soil thermal regime. *J Environ Manage*. 154:166–176. doi:10.1016/j.jenvman.2015.02.037.

Bubier J, Costello A, Moore TR, Roulet NT, Savage K. 1993. Microtopography and methane flux in boreal peatlands, northern Ontario, Canada. *Can J Bot.* 71(8):1056–1063. doi:10.1139/b93-122.

Clemmensen KE, Bahr A, Ovaskainen O, Dahlberg A, Ekblad A, Wallander H, Stenlid J, Finlay RD, Wardle DA, Lindahl BD. 2013. Roots and Associated Fungi Drive Long-Term Carbon Sequestration in Boreal Forest. *Science.* 339(6127):1615–1618. doi:10.1126/science.1231923.

Clymo. 1984. The Limits to Peat Bog Growth. *Philos Trans R Soc Lond.* 303(1117):605–654.

Cody WJ, Wagner V. 1980. The Biology of Canadian Weeds. 49. *Equisitum arvense* L. *Can J Plant Sci.* 61:123–133.

Comas X, Slater L, Reeve AS. 2011. Atmospheric pressure drives changes in the vertical distribution of biogenic free-phase gas in a northern peatland. *J Geophys Res.* 116(G4):G04014. doi:10.1029/2011JG001701.

Daou L, Garnier É, Shipley B. 2021. Quantifying the relationship linking the community-weighted means of plant traits and soil fertility. *Ecology.* 102(9):e03454. doi:10.1002/ecy.3454.

Davidson SJ, Goud EM, Malhotra A, Estey CO, Korsah P, Strack M. 2021. Linear Disturbances Shift Boreal Peatland Plant Communities Toward Earlier Peak Greenness. *J Geophys Res Biogeosciences.* 126(8):e2021JG006403. doi:10.1029/2021JG006403.

Davidson SJ, Van Beest C, Petrone R, Strack M. 2019. Wildfire overrides hydrological controls on boreal peatland methane emissions. *Biogeosciences.* 16(13):2651–2660. doi:10.5194/bg-16-2651-2019.

Davies GM, Gray A, Power SC, Domènech R. 2023. Resilience of temperate peatland vegetation communities to wildfire depends upon burn severity and pre-fire species composition. *Ecol Evol.* 13(4):e9912. doi:10.1002/ece3.9912.

Dinsmore KJ, Skiba UM, Billett MF, Rees RM. 2009. Effect of water table on greenhouse gas emissions from peatland mesocosms. *Plant Soil.* 318(1–2):229–242. doi:10.1007/s11104-008-9832-9.

Dunfield P, Knowles R, Dumont R, Moore T. 1993. Methane production and consumption in temperate and subarctic peat soils: Response to temperature and pH. *Soil Biol Biochem.* 25(3):321–326. doi:10.1016/0038-0717(93)90130-4.

Elmes MC, Kessel E, Wells CM, Sutherland G, Price JS, Macrae ML, Petrone RM. 2021. Evaluating the hydrological response of a boreal fen following the removal of a temporary access road. *J Hydrol.* 594:125928. doi:10.1016/j.jhydrol.2020.125928.

Elmes MC, Thompson DK, Price JS. 2019. Changes to the hydrophysical properties of upland and riparian soils in a burned fen watershed in the Athabasca Oil Sands Region, northern Alberta, Canada. *CATENA.* 181:104077. doi:10.1016/j.catena.2019.104077.

Enright NJ, Fontaine JB, Lamont BB, Miller BP, Westcott VC. 2014. Resistance and resilience to changing climate and fire regime depend on plant functional traits. Cornelissen H, editor. *J Ecol.* 102(6):1572–1581. doi:10.1111/1365-2745.12306.

Eskelinen A, Harrison SP. 2015. Resource colimitation governs plant community responses to altered precipitation. *Proc Natl Acad Sci.* 112(42):13009–13014. doi:10.1073/pnas.1508170112.

Filippa G, Cremonese E, Migliavacca M, Galvagno M, Forkel M, Wingate L, Tomelleri E, Morra Di Cella U, Richardson AD. 2016. Phenopix: A R package for image-based vegetation phenology. *Agric For Meteorol.* 220:141–150. doi:10.1016/j.agrformet.2016.01.006.

French NHF, Kasischke ES, Williams DG. 2002. Variability in the emission of carbon-based trace gases from wildfire in the Alaskan boreal forest. *J Geophys Res Atmospheres.* 107(D1). doi:10.1029/2001JD000480. [accessed 2023 Dec 28].
<https://agupubs.onlinelibrary.wiley.com/doi/10.1029/2001JD000480>.

Gorham E. 1991. Northern Peatlands: Role in the Carbon Cycle and Probable Responses to Climatic Warming. *Ecol Appl.* 1(2):182–195. doi:10.2307/1941811.

Goud EM, Moore TR, Roulet NT. 2017 Sep. Predicting peatland carbon fluxes from non-destructive plant traits. :1824–1833. [accessed 2024 Jun 7].
<https://besjournals.onlinelibrary.wiley.com/doi/10.1111/1365-2435.12891>.

Grau-Andrés R, Davies GM, Waldron S, Scott EM, Gray A. 2019. Increased fire severity alters initial vegetation regeneration across Calluna-dominated ecosystems. *J Environ Manage.* 231:1004–1011. doi:10.1016/j.jenvman.2018.10.113.

Guêné-Nanchen M, LeBlanc M-C, Rochefort L. 2022. Post-fire peatland vegetation recovery: a case study in open rich fens of the Canadian boreal forest. *Botany.* 100(5):435–447. doi:10.1139/cjb-2021-0194.

Hantson S, Arneth A, Harrison SP, Kelley DI, Prentice IC, Rabin SS, Archibald S, Mouillot F, Arnold SR, Artaxo P, et al. 2016. The status and challenge of global fire modelling. *Biogeosciences.* 13(11):3359–3375. doi:10.5194/bg-13-3359-2016.

Harden JW, Trumbore SE, Stocks BJ, Hirsch A, Gower ST, O’neill KP, Kasischke ES. 2000. The role of fire in the boreal carbon budget. *Glob Change Biol.* 6(S1):174–184. doi:10.1046/j.1365-2486.2000.06019.x.

Hashimoto H, Nemani RR, White MA, Jolly WM, Piper SC, Keeling CD, Myneni RB, Running SW. 2004. El Niño–Southern Oscillation–induced variability in terrestrial carbon cycling. *J Geophys Res Atmospheres.* 109(D23):2004JD004959. doi:10.1029/2004JD004959.

Hobbs NB. 1986. Mire morphology and the properties and behaviour of some British and foreign peats. *Q J Eng Geol.* 19(1):7–80. doi:10.1144/GSL.QJEG.1986.019.01.02.

Hu Y, Christensen EG, Amin HMF, Smith TEL, Rein G. 2019. Experimental study of moisture content effects on the transient gas and particle emissions from peat fires. *Combust Flame*. 209:408–417. doi:10.1016/j.combustflame.2019.07.046.

Hu Y, Fernandez-Anez N, Smith TEL, Rein G. 2018. Review of emissions from smouldering peat fires and their contribution to regional haze episodes. *Int J Wildland Fire*. 27(5):293. doi:10.1071/WF17084.

Huber JA, May K, Hülsbergen K-J. 2017. Allometric tree biomass models of various species grown in short-rotation agroforestry systems. *Eur J For Res*. 136(1):75–89. doi:10.1007/s10342-016-1010-7.

van Hulzen JB, Segers R, van Bodegom PM, Leffelaar PA. 1999. Temperature effects on soil methane production: an explanation for observed variability.

Intergovernmental Panel On Climate Change. 2023. *Climate Change 2021 – The Physical Science Basis: Working Group I Contribution to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*. 1st ed. Cambridge University Press. [accessed 2024 Feb 28]. <https://www.cambridge.org/core/product/identifier/9781009157896/type/book>.

Jassey VEJ, Reczuga MK, Zielińska M, Słowińska S, Robroek BJM, Mariotte P, Seppey CVW, Lara E, Barabach J, Słowiński M, et al. 2018 Mar. Tipping point in plant–fungal interactions under severe drought causes abrupt rise in peatland ecosystem respiration. *Glob Change Biol*. [accessed 2024 Apr 26]. <https://onlinelibrary.wiley.com/doi/10.1111/gcb.13928>.

Johnson D, Kershaw L, MacKinnon A, Pojar J. 2017. *Plants of the Western Forest: Alaska to Minnesota Boreal and Aspen Parkland*. Edmonton, Alberta: Lone Pine.

Kane ES, Veverica TJ, Tfaily MM, Lilleskov EA, Meingast KM, Kolka RK, Daniels AL, Chimner RA. 2019. Reduction-Oxidation Potential and Dissolved Organic Matter Composition in Northern Peat Soil: Interactive Controls of Water Table Position and Plant Functional Groups. *J Geophys Res Biogeosciences*. 124(11):3600–3617. doi:10.1029/2019JG005339.

Kechavarzi C, Dawson Q, Bartlett M, Leeds-Harrison PB. 2010. The role of soil moisture, temperature and nutrient amendment on CO₂ efflux from agricultural peat soil microcosms. *Geoderma*. 154(3–4):203–210. doi:10.1016/j.geoderma.2009.02.018.

Kelly R, Boston E, Montgomery WI, Reid N. 2016. The role of the seed bank in recovery of temperate heath and blanket bog following wildfires. Marrs R, editor. *Appl Veg Sci*. 19(4):620–633. doi:10.1111/avsc.12242.

Kettridge N, Humphrey RE, Smith JE, Lukenbach MC, Devito KJ, Petrone RM, Waddington JM. 2014. Burned and unburned peat water repellency: Implications for peatland evaporation following wildfire. *J Hydrol*. 513:335–341. doi:10.1016/j.jhydrol.2014.03.019.

Kettridge N, Turetsky MR, Sherwood JH, Thompson DK, Miller CA, Benscoter BW, Flannigan MD, Wotton BM, Waddington JM. 2015. Moderate drop in water table increases peatland vulnerability to post-fire regime shift. *Sci Rep.* 5(1):8063. doi:10.1038/srep08063.

Kleinke K, Davidson SJ, Schmidt M, Xu B, Strack M. 2022. How mounds are made matters: seismic line restoration techniques affect peat physical and chemical properties throughout the peat profile. *Can J For Res.* 52(6):963–976. doi:10.1139/cjfr-2022-0015.

Kohlenberg AJ, Turetsky MR, Thompson DK, Branfireun BA, Mitchell CPJ. 2018. Controls on boreal peat combustion and resulting emissions of carbon and mercury. *Environ Res Lett.* 13(3):035005. doi:10.1088/1748-9326/aa9ea8.

Konings W, Boyd KG, Andersen R. 2019. Comparison of plant traits of sedges, shrubs and Sphagnum mosses between sites undergoing forest-to-bog restoration and near-natural open blanket bog: a pilot study. *Mires Peat.*(23):1–10. doi:10.19189/MaP.2017.OMB.307.

Kotowski W, Jabłońska E, Bartoszek H. 2013. Conservation management in fens: Do large tracked mowers impact functional plant diversity? *Biol Conserv.* 167:292–297. doi:10.1016/j.biocon.2013.08.021.

Kurth JM, Smit NT, Berger S, Schouten S, Jetten MSM, Welte CU. 2019. Anaerobic methanotrophic archaea of the ANME-2d clade feature lipid composition that differs from other ANME archaea. *FEMS Microbiol Ecol.* 95(7):fiz082. doi:10.1093/femsec/fiz082.

Lai DYF. 2009. Methane dynamics in northern peatlands: A review. *Pedosphere.* 19(4):409–421.

Laine AM, Korrensalo A, Kokkonen NAK, Tuittila E. 2021. Impact of long-term water level drawdown on functional plant trait composition of northern peatlands. *Funct Ecol.* 35(10):2342–2357. doi:10.1111/1365-2435.13883.

Laine AM, Korrensalo A, Tuittila E-S. 2022. Plant functional traits play the second fiddle to plant functional types in explaining peatland CO₂ and CH₄ gas exchange. *Sci Total Environ.* 834:155352. doi:10.1016/j.scitotenv.2022.155352.

Laine AM, Mäkiranta P, Laiho R, Mehtätalo L, Penttilä T, Korrensalo A, Minkkinen K, Fritze H, Tuittila E. 2019. Warming impacts on boreal fen CO₂ exchange under wet and dry conditions. *Glob Change Biol.* 25(6):1995–2008. doi:10.1111/gcb.14617.

Laine AM, Selänpää T, Oksanen J, Seväkivi M, Tuittila E-S. 2018. Plant diversity and functional trait composition during mire development. *Mires Peat.*(21):1–19. doi:10.19189/MaP.2017.OMB.280.

Lasslop G, Coppola AI, Voulgarakis A, Yue C, Veraverbeke S. 2019. Influence of Fire on the Carbon Cycle and Climate. *Curr Clim Change Rep.* 5(2):112–123. doi:10.1007/s40641-019-00128-9.

- Li F, Lawrence DM, Bond-Lamberty B. 2017. Impact of fire on global land surface air temperature and energy budget for the 20th century due to changes within ecosystems. *Environ Res Lett.* 12(4):044014. doi:10.1088/1748-9326/aa6685.
- Li Q, Gogo S, Leroy F, Guimbaud C, Laggoun-Défarge F. 2021. Response of Peatland CO₂ and CH₄ Fluxes to Experimental Warming and the Carbon Balance. *Front Earth Sci.* 9:631368. doi:10.3389/feart.2021.631368.
- Li Z, Wei J, Zhou X, Tian Q, He W, Cao X. 2024. Dynamic restoration mechanism of plant community in the burned area of northeastern margin of Qinghai-Tibet Plateau. *Front Plant Sci.* 15:1368814. doi:10.3389/fpls.2024.1368814.
- Limpens J, Berendse F, Blodau C, Canadell JG, Freeman C, Holden J, Roulet N, Rydin H, Schaepman-Strub G. 2008. Peatlands and the carbon cycle: from local processes to global implications – a synthesis. *Biogeosciences.* 5(5):1475–1491. doi:10.5194/bg-5-1475-2008.
- Loisel J, Yu Z, Beilman DW, Camill P, Alm J, Amesbury MJ, Anderson D, Andersson S, Bochicchio C, Barber K, et al. 2014. A database and synthesis of northern peatland soil properties and Holocene carbon and nitrogen accumulation. *The Holocene.* 24(9):1028–1042. doi:10.1177/0959683614538073.
- Lukenbach MC, Devito KJ, Kettridge N, Petrone RM, Waddington JM. 2016. Burn severity alters peatland moss water availability: implications for post-fire recovery. *Ecohydrology.* 9(2):341–353. doi:10.1002/eco.1639.
- Martin JL, Gower ST, Plaut J, Holmes B. 2005. Carbon pools in a boreal mixedwood logging chronosequence. *Glob Change Biol.* 11(11):1883–1894. doi:10.1111/j.1365-2486.2005.01019.x.
- McCarter CPR, Price JS. 2014. Ecohydrology of *Sphagnum* moss hummocks: mechanisms of capitula water supply and simulated effects of evaporation. *Ecohydrology.* 7(1):33–44. doi:10.1002/eco.1313.
- Mikita-Barbato RA, Kelly JJ, Tate RL. 2015. Wildfire effects on the properties and microbial community structure of organic horizon soils in the New Jersey Pinelands. *Soil Biol Biochem.* 86:67–76. doi:10.1016/j.soilbio.2015.03.021.
- Moore TR, Bubier JL, Frolking SE, Lafleur PM, Roulet NT. 2002. Plant biomass and production and CO₂ exchange in an ombrotrophic bog. *J Ecol.* 90(1):25–36. doi:10.1046/j.0022-0477.2001.00633.x.
- Moore TR, Dalva M. 1997. Methane and carbon dioxide exchange potentials of peat soils in aerobic and anaerobic laboratory incubations. *Soil Biol Biochem.* 29(8):1157–1164.
- Moore TR, De Young A, Bubier JL, Humphreys ER, Lafleur PM, Roulet NT. 2011. A Multi-Year Record of Methane Flux at the Mer Bleue Bog, Southern Canada. *Ecosystems.* 14(4):646–657. doi:10.1007/s10021-011-9435-9.

- Morison M, Van Beest C, Macrae M, Nwaishi F, Petrone R. 2021. Deeper burning in a boreal fen peatland 1-year post-wildfire accelerates recovery trajectory of carbon dioxide uptake. *Ecohydrology*. 14(3):e2277. doi:10.1002/eco.2277.
- Nelson K, Thompson D, Hopkinson C, Petrone R, Chasmer L. 2021. Peatland-fire interactions: A review of wildland fire feedbacks and interactions in Canadian boreal peatlands. *Sci Total Environ*. 769:145212. doi:10.1016/j.scitotenv.2021.145212.
- Orwin KH, Ostle NJ. 2012. Moss species effects on peatland carbon cycling after fire. *Funct Ecol*. 26(4):829–836. doi:10.1111/j.1365-2435.2012.01991.x.
- Pausas JG, Keeley JE. 2014. Evolutionary ecology of resprouting and seeding in fire-prone ecosystems. *New Phytol*. 204(1):55–65. doi:10.1111/nph.12921.
- Pearce DME, Clymo RS. 2001. Methane oxidation in a peatland core. *Glob Biogeochem Cycles*. 15(3):709–720. doi:10.1029/2000GB001323.
- Pekin BK, Wittkuhn RS, Boer MM, Macfarlane C, Grierson PF. 2011. Plant functional traits along environmental gradients in seasonally dry and fire-prone ecosystem: Plant functional traits along environment gradients. *J Veg Sci*. 22(6):1009–1020. doi:10.1111/j.1654-1103.2011.01323.x.
- Pinheiro J, Bates D. 2000. *Mixed-Effects Models in S and S-PLUS*. New York: Springer-Verlag (Statistics and Computing). [accessed 2024 Sep 3]. <http://link.springer.com/10.1007/b98882>.
- R Core Team (2022). *R: A language and environment for statistical computing*. R Foundation for Statistical Computing, Vienna, Austria. URL <https://www.R-project.org/>.
- Rogers BM, Soja AJ, Goulden ML, Randerson JT. 2015. Influence of tree species on continental differences in boreal fires and climate feedbacks. *Nat Geosci*. 8(3):228–234. doi:10.1038/ngeo2352.
- Roulet N, Moore T, Bubier J, Lafleur P. 1991. Northern fens: methane flux and climactic change. *Tellus*. 1992(44B):100–105.
- Santín C, Doerr SH, Kane ES, Masiello CA, Ohlson M, De La Rosa JM, Preston CM, Dittmar T. 2016. Towards a global assessment of pyrogenic carbon from vegetation fires. *Glob Change Biol*. 22(1):76–91. doi:10.1111/gcb.12985.
- Schlesinger WH, Andrews JA. 2000. Soil respiration and the global carbon cycle. *Biogeochemistry*. 48(1):7–20. doi:10.1023/A:1006247623877.
- Schneider CA, Rasband WS, Eliceiri KW. 2012. NIH Image to ImageJ: 25 years of image analysis. *Nat Methods*. 9(7):671–675. doi:10.1038/nmeth.2089.
- Segers R. 1998. [No title found]. *Biogeochemistry*. 41(1):23–51. doi:10.1023/A:1005929032764.

Shepherd HER, Catford JA, Steele MN, Dumont MG, Mills RTE, Hughes PDM, Robroek BJM. 2021. Propagule availability drives post-wildfire recovery of peatland plant communities. Alday J, editor. Appl Veg Sci. 24(3):e12608. doi:10.1111/avsc.12608.

Smart SM, Glanville HC, Blanes MDC, Mercado LM, Emmett BA, Jones DL, Cosby BJ, Marrs RH, Butler A, Marshall MR, et al. 2017. Leaf dry matter content is better at predicting above-ground net primary production than specific leaf area. Field K, editor. Funct Ecol. 31(6):1336–1344. doi:10.1111/1365-2435.12832.

St James AR, Yavitt JB, Zinder SH, Richardson RE. 2021. Linking microbial *Sphagnum* degradation and acetate mineralization in acidic peat bogs: from global insights to a genome-centric case study. ISME J. 15(1):293–303. doi:10.1038/s41396-020-00782-0.

Stalheim T, Ballance S, Christensen BE, Granum PE. 2009. Sphagnan - a pectin-like polymer isolated from *Sphagnum* moss can inhibit the growth of some typical food spoilage and food poisoning bacteria by lowering the pH. J Appl Microbiol. 106(3):967–976. doi:10.1111/j.1365-2672.2008.04057.x.

Stocks BJ, Mason JA, Todd JB, Bosch EM, Wotton BM, Amiro BD, Flannigan MD, Hirsch KG, Logan KA, Martell DL, et al. 2002. Large forest fires in Canada, 1959–1997. J Geophys Res Atmospheres. 107(D1). doi:10.1029/2001JD000484. [accessed 2024 Apr 24]. <https://agupubs.onlinelibrary.wiley.com/doi/10.1029/2001JD000484>.

Strack M, Waddington JM, Bourbonniere RA, Buckton EL, Shaw K, Whittington P, Price JS. 2008. Effect of water table drawdown on peatland dissolved organic carbon export and dynamics. Hydrol Process. 22(17):3373–3385. doi:10.1002/hyp.6931.

Street LE, Subke J-A, Baxter R, Dinsmore KJ, Knoblauch C, Wookey PA. 2018. Ecosystem carbon dynamics differ between tundra shrub types in the western Canadian Arctic. Environ Res Lett. 13(8):084014. doi:10.1088/1748-9326/aad363.

Sulman BN, Desai AR, Saliendra NZ, Lafleur PM, Flanagan LB, Sonnentag O, Mackay DS, Barr AG, Van Der Kamp G. 2010. CO₂ fluxes at northern fens and bogs have opposite responses to inter-annual fluctuations in water table. Geophys Res Lett. 37(19):2010GL044018. doi:10.1029/2010GL044018.

Sumner DM. 2007. Effects of capillarity and microtopography on wetland specific yield. Wetlands. 27(3):693–701. doi:10.1672/0277-5212(2007)27[693:EOCAMO]2.0.CO;2.

Susilawati HL, Ariani M, Nurhasan, Setyanto P. 2022. Response of temperature, moisture and CO₂ emission from different water levels at undisturbed peat soil column. IOP Conf Ser Earth Environ Sci. 1025(1):012028. doi:10.1088/1755-1315/1025/1/012028.

Swails EE, Ardón M, Krauss KW, Peralta AL, Emanuel RE, Helton AM, Morse JL, Gutenberg L, Cormier N, Shoch D, et al. 2022. Response of soil respiration to changes in soil temperature and

water table level in drained and restored peatlands of the southeastern United States. *Carbon Balance Manag.* 17(1):18. doi:10.1186/s13021-022-00219-5.

Tarnocai C. 2006. The effect of climate change on carbon in Canadian peatlands. *Glob Planet Change.* 53(4):222–232. doi:10.1016/j.gloplacha.2006.03.012.

Thompson DK, Benscoter BW, Waddington JM. 2014. Water balance of a burned and unburned forested boreal peatland. *Hydrol Process.* 28(24):5954–5964. doi:10.1002/hyp.10074.

Thompson DK, Waddington JM. 2013. Peat properties and water retention in boreal forested peatlands subject to wildfire: Peat Properties in Fire-Affected Peatlands. *Water Resour Res.* 49(6):3651–3658. doi:10.1002/wrcr.20278.

Tolunay D, Kowalchuk GA, Erkens G, Hefting MM. 2024. Aerobic and anaerobic decomposition rates in drained peatlands: Impact of botanical composition. *Sci Total Environ.* 930:172639. doi:10.1016/j.scitotenv.2024.172639.

Toomey M, Friedl MA, Frolking S, Hufkens K, Klosterman S, Sonnentag O, Baldocchi DD, Bernacchi CJ, Biraud SC, Bohrer G, et al. 2015. Greenness indices from digital cameras predict the timing and seasonal dynamics of canopy-scale photosynthesis. *Ecol Appl.* 25(1):99–115. doi:10.1890/14-0005.1.

Treat CC, Bubier JL, Varner RK, Crill PM. 2007. Timescale dependence of environmental and plant-mediated controls on CH₄ flux in a temperate fen. *J Geophys Res Biogeosciences.* 112(G1):2006JG000210. doi:10.1029/2006JG000210.

Treat CC, Wollheim WM, Varner RK, Grandy AS, Talbot J, Frolking S. 2014. Temperature and peat type control CO₂ and CH₄ production in Alaskan permafrost peats. *Glob Change Biol.* 20(8):2674–2686. doi:10.1111/gcb.12572.

Turetsky MR, Amiro BD, Bosch E, Bhatti JS. 2004. Historical burn area in western Canadian peatlands and its relationship to fire weather indices. *Glob Biogeochem Cycles.* 18(4):2004GB002222. doi:10.1029/2004GB002222.

Turetsky MR, Benscoter B, Page S, Rein G, Van Der Werf GR, Watts A. 2015. Global vulnerability of peatlands to fire and carbon loss. *Nat Geosci.* 8(1):11–14. doi:10.1038/ngeo2325.

Turetsky MR, Kane ES, Harden JW, Ottmar RD, Manies KL, Hoy E, Kasischke ES. 2011. Recent acceleration of biomass burning and carbon losses in Alaskan forests and peatlands. *Nat Geosci.* 4(1):27–31. doi:10.1038/ngeo1027.

Turetsky MR, Kotowska A, Bubier J, Dise NB, Crill P, Hornibrook ERC, Minkinen K, Moore TR, Myers-Smith IH, Nykänen H, et al. 2014. A synthesis of methane emissions from 71 northern, temperate, and subtropical wetlands. *Glob Change Biol.* 20(7):2183–2197. doi:10.1111/gcb.12580.

- Van Beest C, Petrone R, Nwaishi F, Waddington JM, Macrae M. 2019. Increased Peatland Nutrient Availability Following the Fort McMurray Horse River Wildfire. *Diversity*. 11(9):142. doi:10.3390/d11090142.
- Van Der Werf GR, Randerson JT, Giglio L, Van Leeuwen TT, Chen Y, Rogers BM, Mu M, Van Marle MJE, Morton DC, Collatz GJ, et al. 2017. Global fire emissions estimates during 1997–2016. *Earth Syst Sci Data*. 9(2):697–720. doi:10.5194/essd-9-697-2017.
- Waddington JM, Morris PJ, Kettridge N, Granath G, Thompson DK, Moore PA. 2015. Hydrological feedbacks in northern peatlands. *Ecohydrology*. 8(1):113–127. doi:10.1002/eco.1493.
- Ward SE, Ostle NJ, Oakley S, Quirk H, Stott A, Henrys PA, Scott WA, Bardgett RD. 2012. Fire Accelerates Assimilation and Transfer of Photosynthetic Carbon from Plants to Soil Microbes in a Northern Peatland. *Ecosystems*. 15(8):1245–1257. doi:10.1007/s10021-012-9581-8.
- Wieder RK, Scott KD, Kamminga K, Vile MA, Vitt DH, Bone T, Xu B, Benscoter BW, Bhatti JS. 2009. Postfire carbon balance in boreal bogs of Alberta, Canada. *Glob Change Biol*. 15(1):63–81. doi:10.1111/j.1365-2486.2008.01756.x.
- Wilkinson SL, Andersen R, Moore PA, Davidson SJ, Granath G, Waddington JM. 2023. Wildfire and degradation accelerate northern peatland carbon release. *Nat Clim Change*. 13(5):456–461. doi:10.1038/s41558-023-01657-w.
- Wilkinson SL, Moore PA, Flannigan MD, Wotton BM, Waddington JM. 2018. Did enhanced afforestation cause high severity peat burn in the Fort McMurray Horse River wildfire? *Environ Res Lett*. 13(1):014018. doi:10.1088/1748-9326/aaa136.
- Wilson PJ, Thompson K, Hodgson JG. 1999. Specific leaf area and leaf dry matter content as alternative predictors of plant strategies. *New Phytol*. 143(1):155–162. doi:10.1046/j.1469-8137.1999.00427.x.
- Yavitt JB, Kryczka AK, Huber ME, Pipes GT, Rodriguez AM. 2019. Inferring Methane Production by Decomposing Tree, Shrub, and Grass Leaf Litter in Bog and Rich Fen Peatlands. *Front Environ Sci*. 7:182. doi:10.3389/fenvs.2019.00182.
- Yu Z, Loisel J, Brosseau DP, Beilman DW, Hunt SJ. 2010. Global peatland dynamics since the Last Glacial Maximum. *Geophys Res Lett*. 37(13):2010GL043584. doi:10.1029/2010GL043584.
- Zhao B, Zhuang Q. 2023. Peatlands and their carbon dynamics in northern high latitudes from 1990 to 2300: a process-based biogeochemistry model analysis. *Biogeosciences*. 20(1):251–270. doi:10.5194/bg-20-251-2023.
- Zoltai SC, Vitt DH. 1995. Canadian wetlands: Environmental gradients and classification. *Vegetatio*. 118(1–2):131–137. doi:10.1007/BF00045195.

Appendix

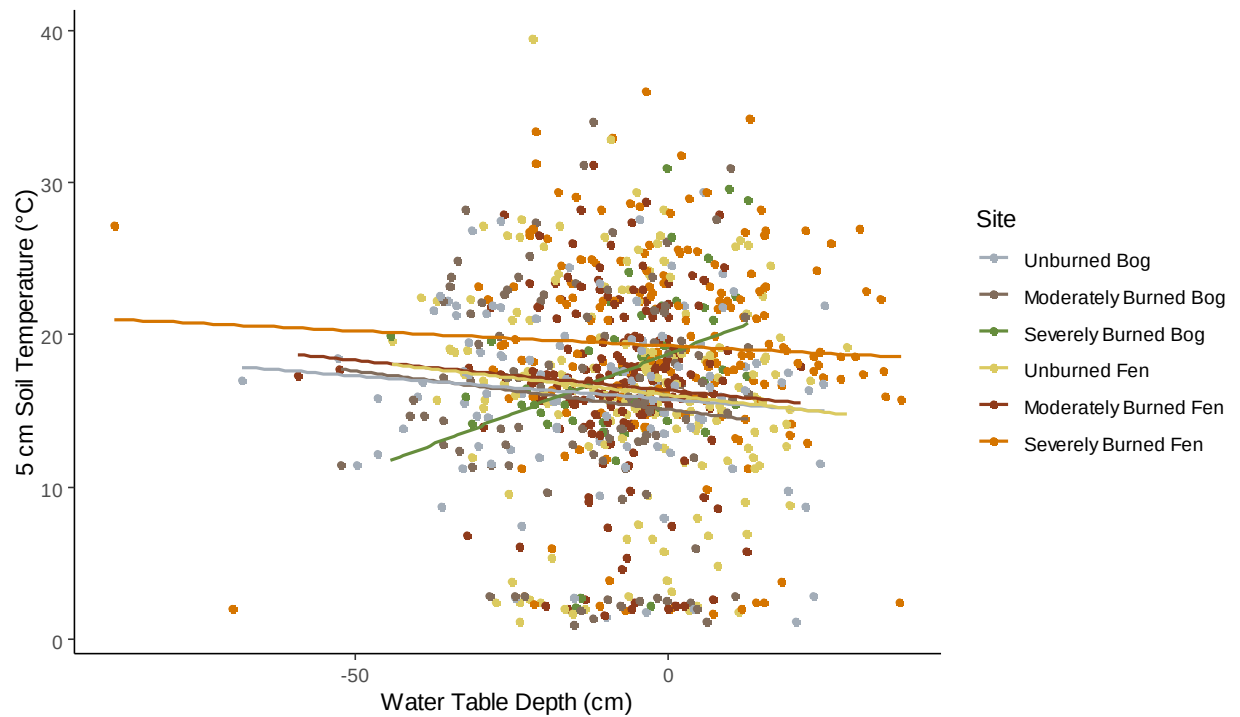


Figure A 1: Scatterplot depicting the variation in 5 cm soil temperature (°C) compared to water table depth (cm). The y-axis represents 5 cm soil temperature (°C), while the x-axis represents water table depth (cm).

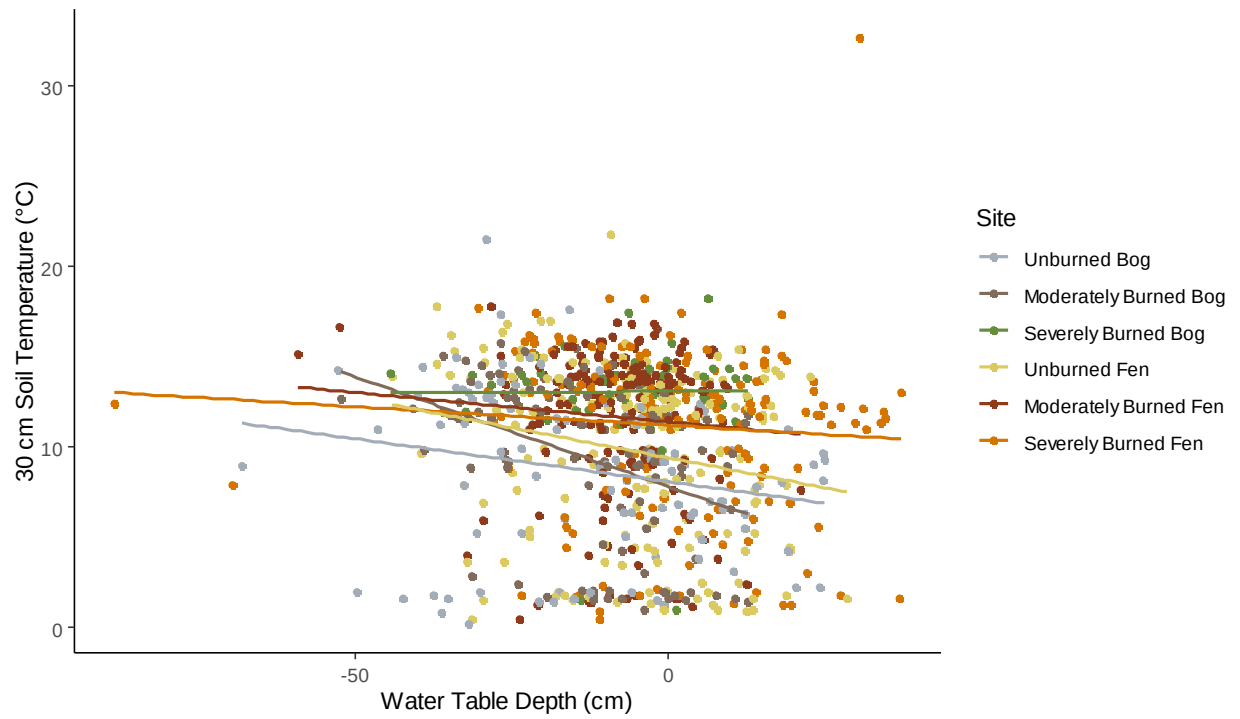


Figure A 2: Scatterplot depicting the variation in 30 cm soil temperature (°C) compared to water table depth (cm). The y-axis represents 30 cm soil temperature (°C), while the x-axis represents water table depth (cm).

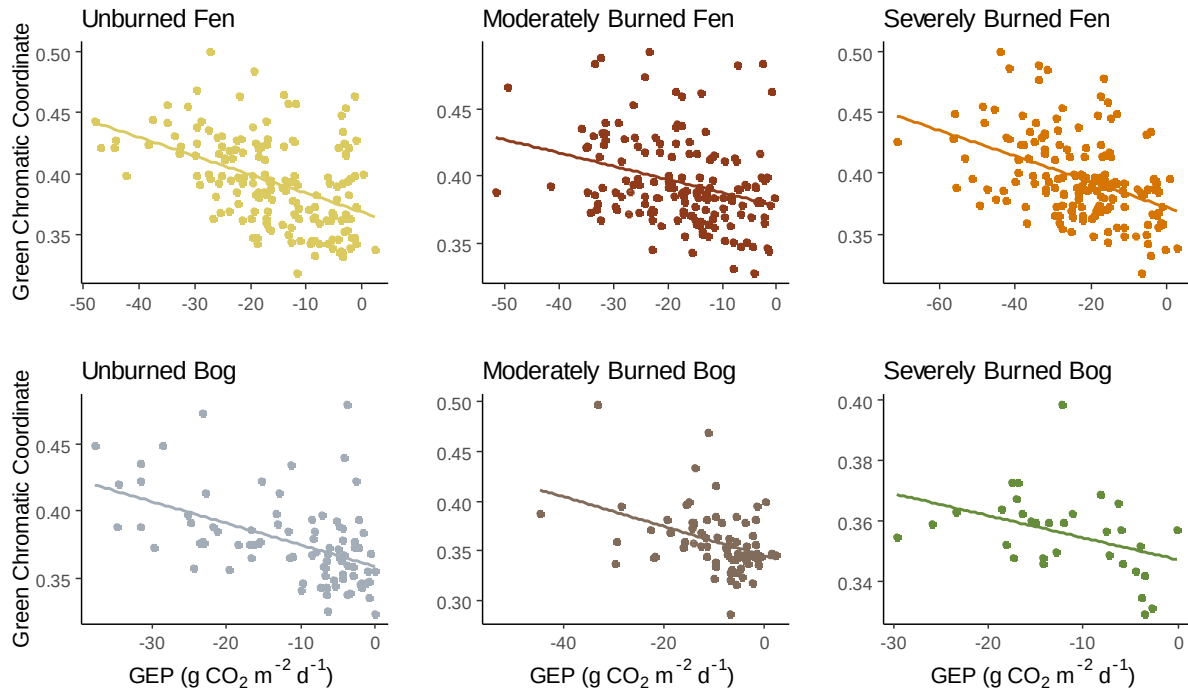


Figure A 3: Scatterplot depicting the relationship between gross ecosystem productivity (GEP) and green chromatic coordinate (GCC). The x-axes represent GEP, while the y-axes represent GCC. Each plot depicts the relationship across the sites.

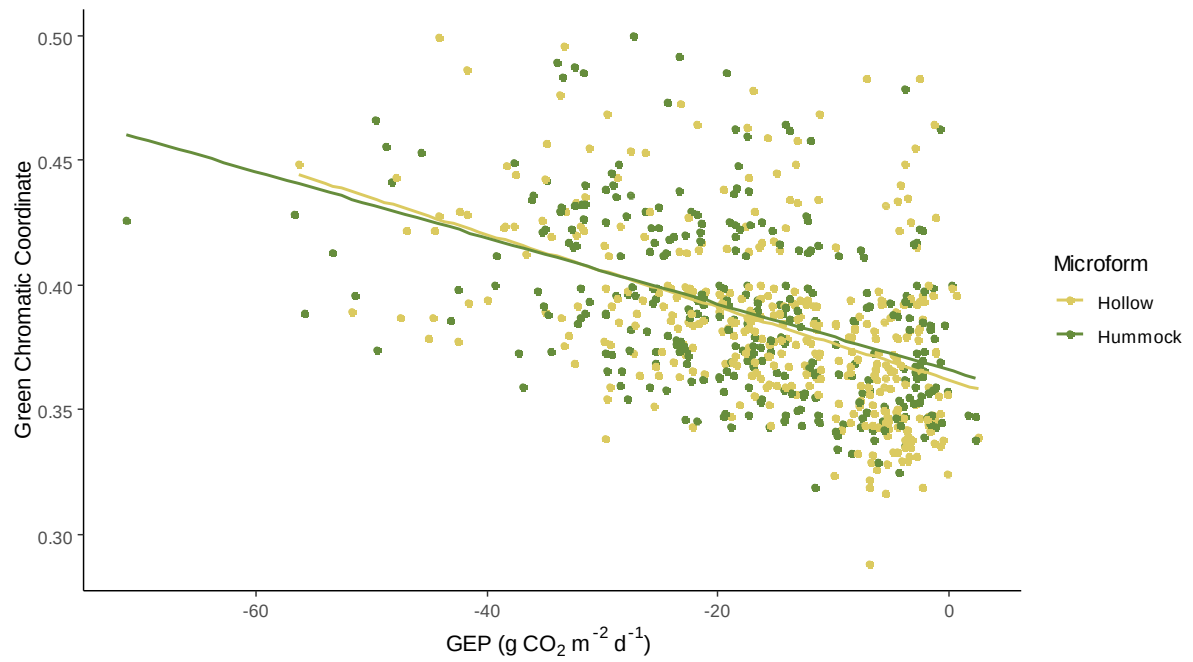


Figure A 4: Scatterplot depicting the relationship between gross ecosystem productivity (GEP) and green chromatic coordinate (GCC) across microforms. The x-axes represent GEP, while the y-axes represent GCC.

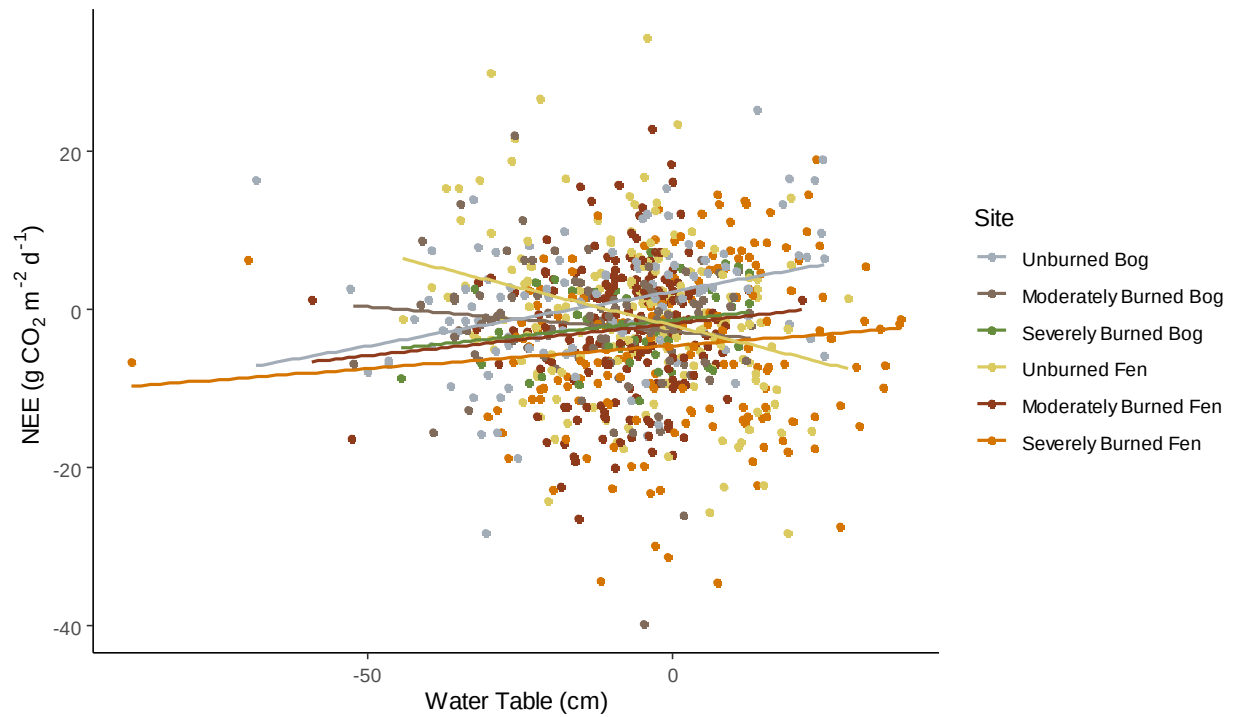


Figure A 5: Scatterplot depicting the variation in Net Ecosystem Exchange (NEE) across different water table depths (cm). The y-axis represents NEE, while the x-axis represents water table depths (cm).

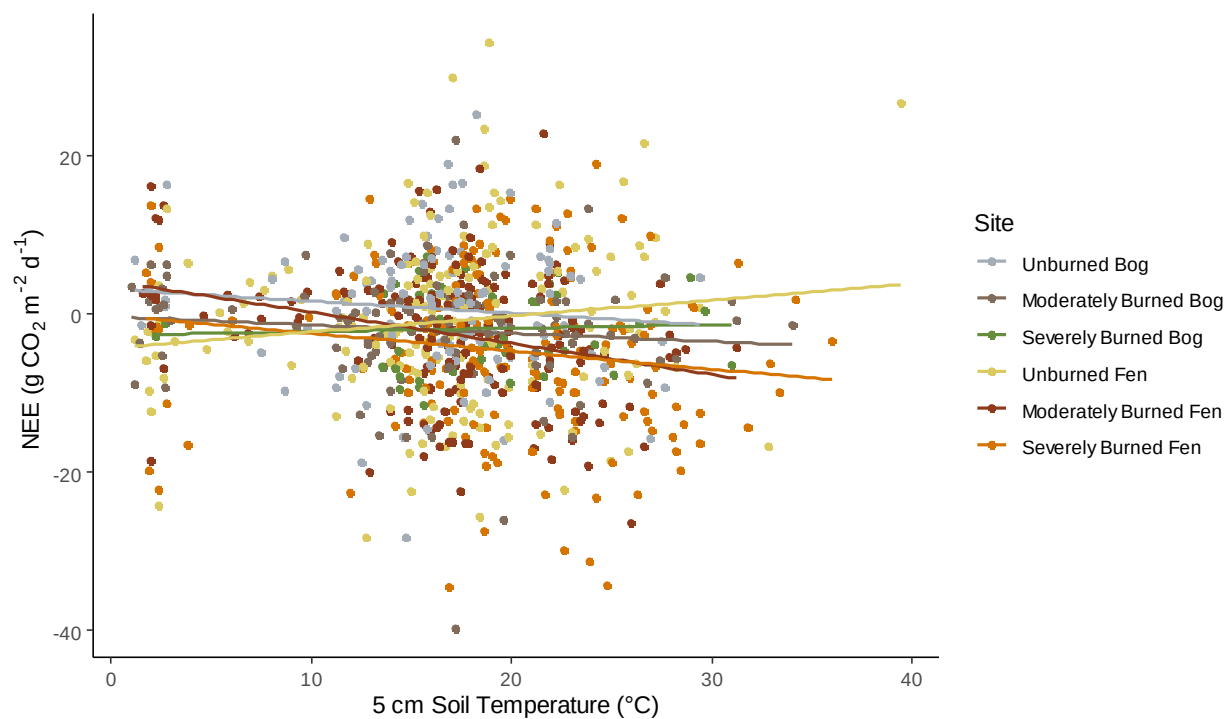


Figure A 6: Scatterplot depicting the variation in Net Ecosystem Exchange (NEE) across 5 cm soil temperatures (°C). The y-axis represents NEE, while the x-axis represents 5 cm soil temperatures (°C).

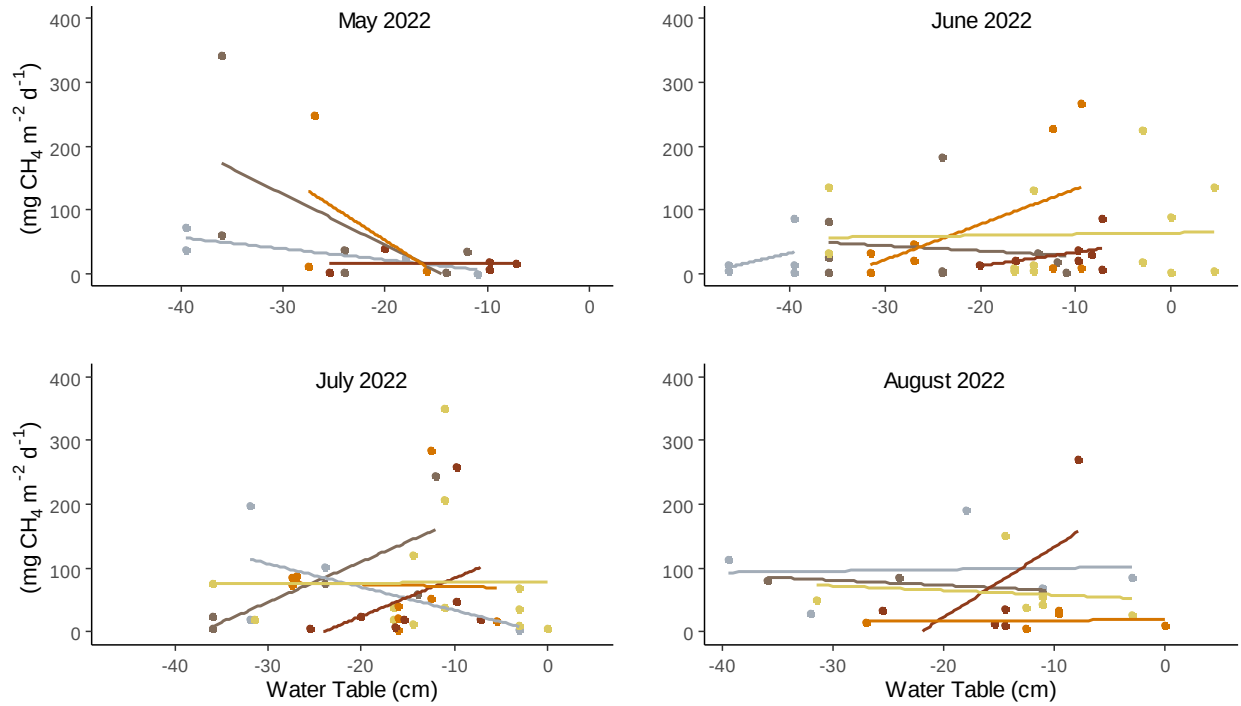


Figure A 7: Scatterplot depicting the variation in CH₄ fluxes across different water table depths (cm) in each month of the sampling period in 2022. The y-axis represents CH₄ fluxes, while the x-axis shows water table depths (cm).

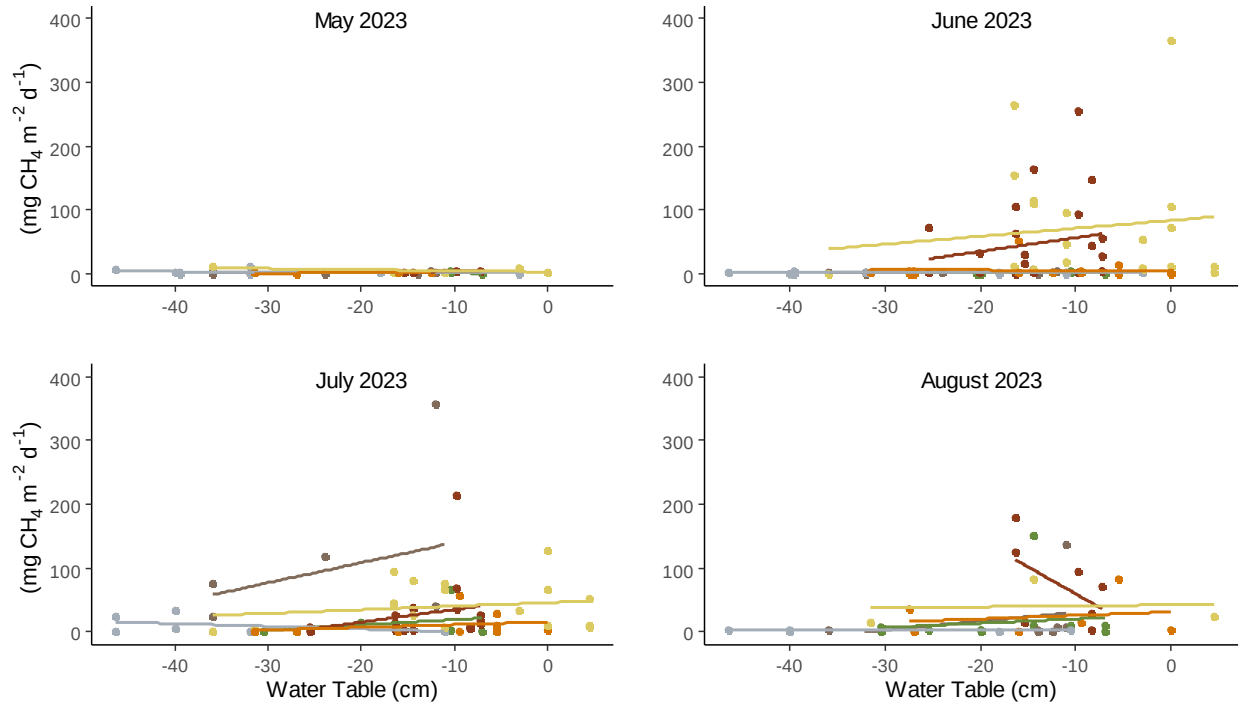


Figure A 8: Scatterplot depicting the variation in CH₄ fluxes across different water table depths (cm) in each month of the sampling period in 2023. The y-axis represents CH₄ fluxes, while the x-axis shows water table depths (cm).

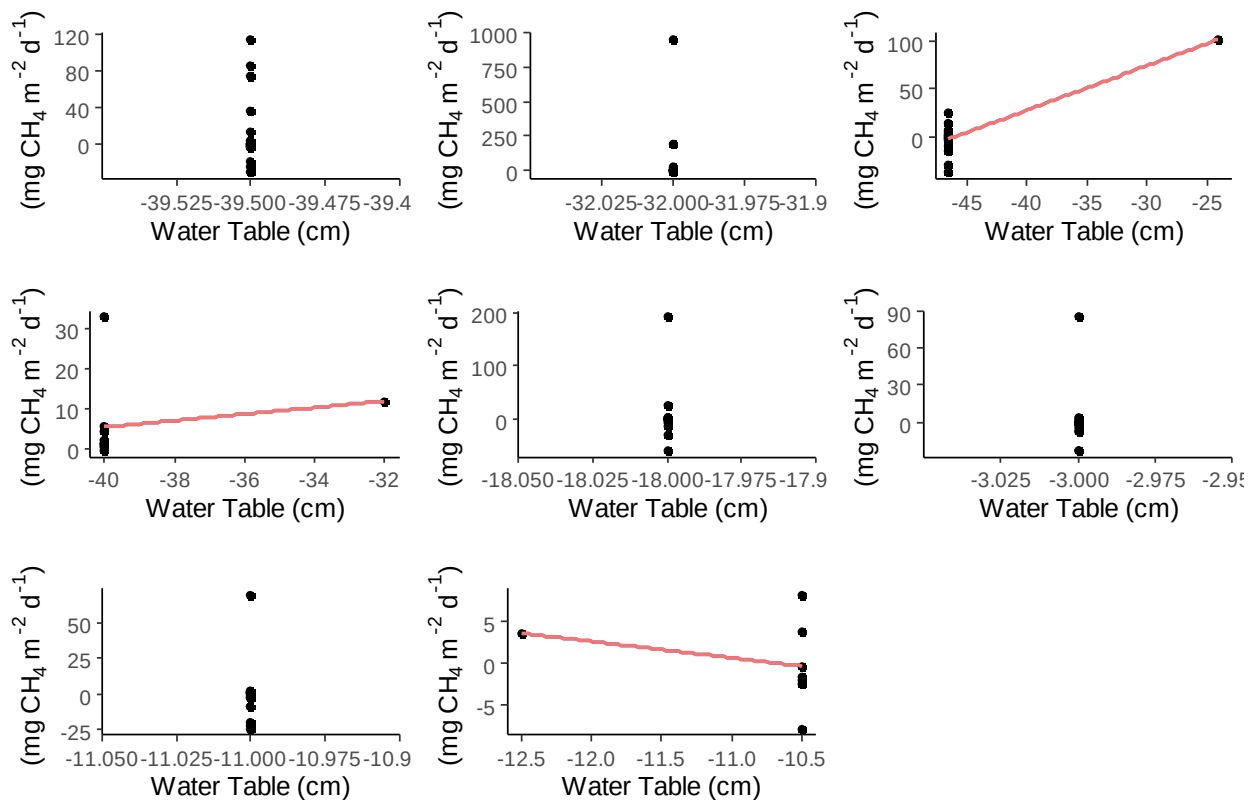


Figure A 9: Scatterplots depicting the variation in CH₄ fluxes across different water table depths (cm) at each of the Unburned Bog's collars. The y-axis represents CH₄ fluxes, while the x-axis shows water table depths (cm). Collars are ordered as such in the figure: top left corner to bottom centre; 1A, 2A, 3A, 4A, 1B, 2B, 3B, 4B.

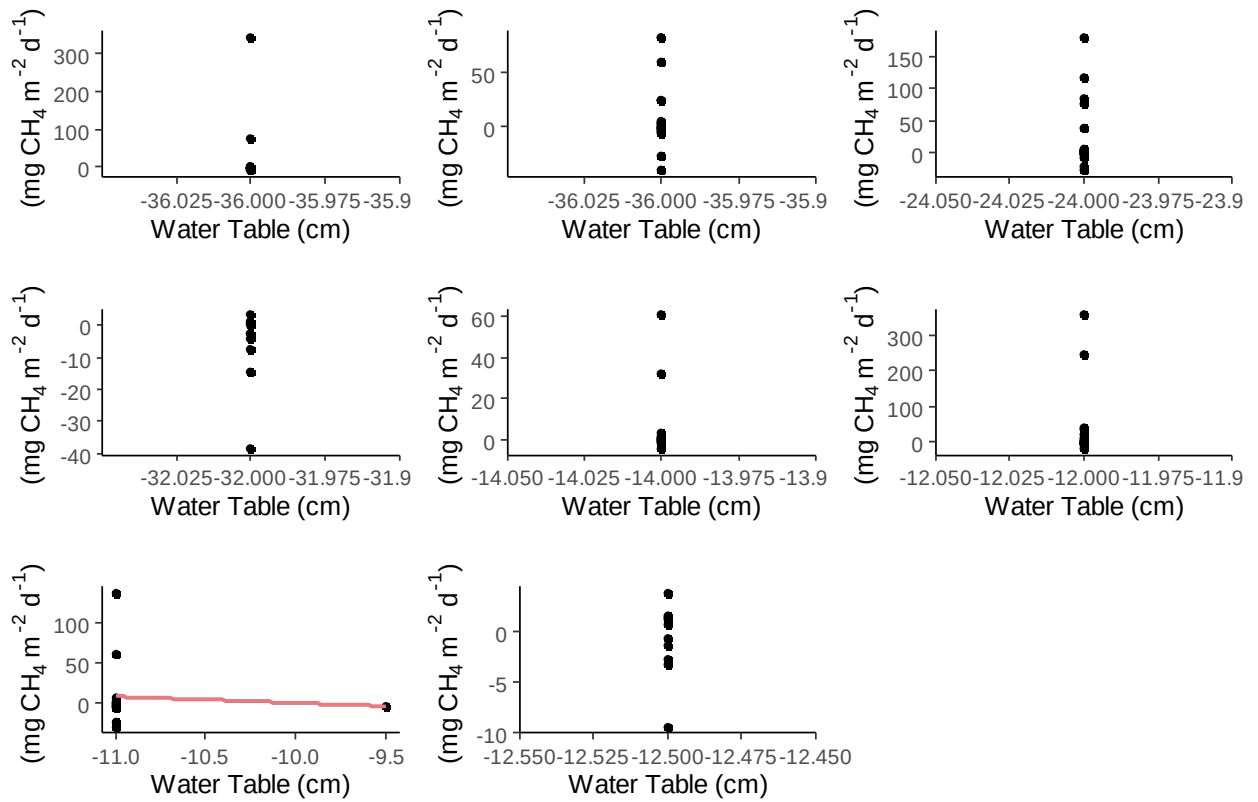


Figure A 10: Scatterplots depicting the variation in CH₄ fluxes across different water table depths (cm) at each of the Moderately Burned Bog's collars. The y-axis represents CH₄ fluxes, while the x-axis shows water table depths (cm). Collars are ordered as such in the figure: top left corner to bottom centre; 1A, 2A, 3A, 4A, 1B, 2B, 3B, 4B.

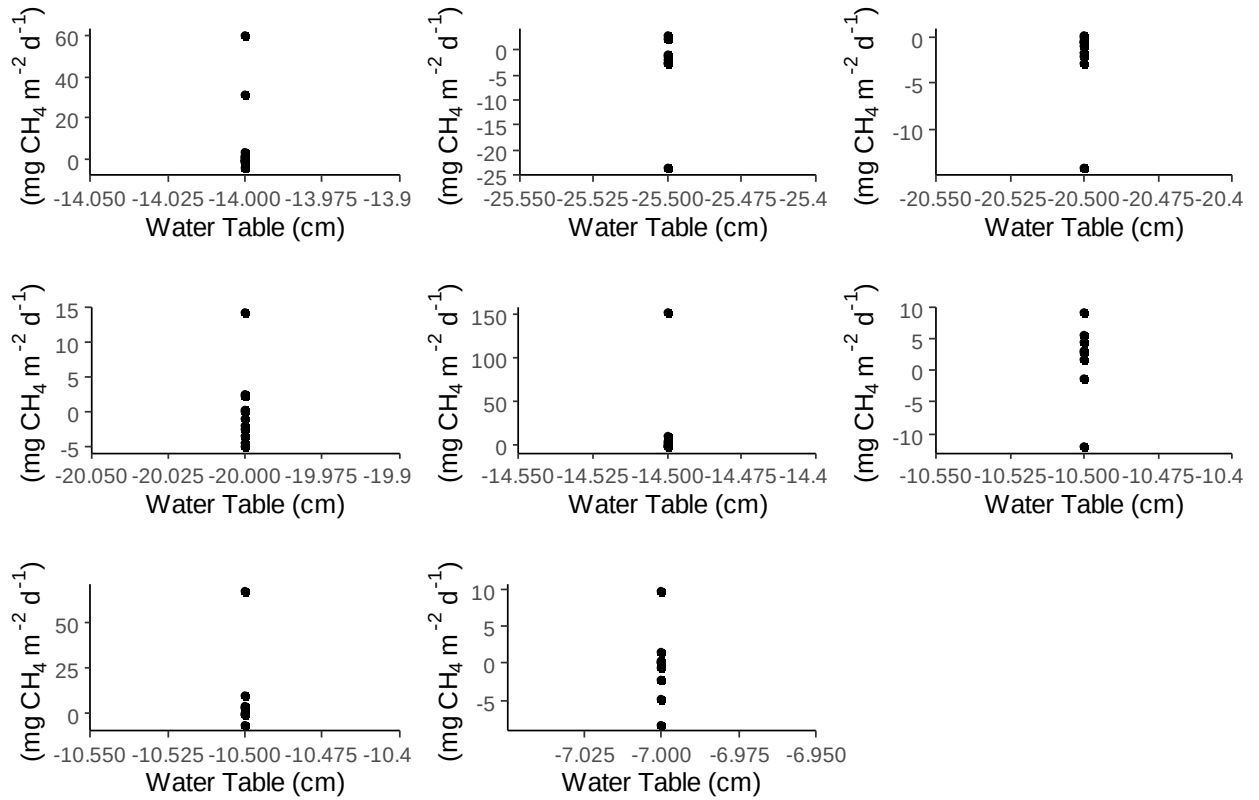


Figure A 11: Scatterplots depicting the variation in CH₄ fluxes across different water table depths (cm) at each of the Severely Burned Bog's collars. The y-axis represents CH₄ fluxes, while the x-axis shows water table depths (cm). Collars are ordered as such in the figure: top left corner to bottom centre; 1A, 2A, 3A, 4A, 1B, 2B, 3B, 4B.

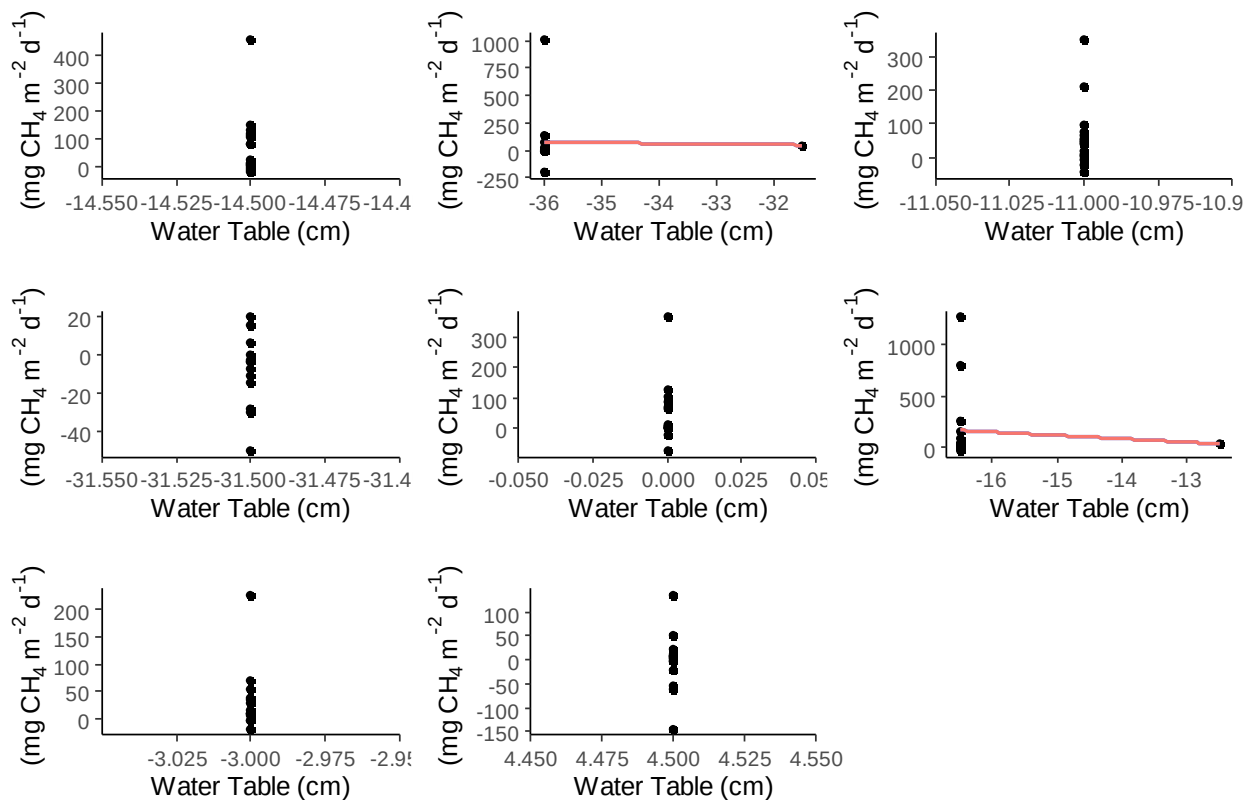


Figure A 12: Scatterplots depicting the variation in CH₄ fluxes across different water table depths (cm) at each of the Unburned Fen's collars. The y-axis represents CH₄ fluxes, while the x-axis shows water table depths (cm). Collars are ordered as such in the figure: top left corner to bottom centre; 1A, 2A, 3A, 4A, 1B, 2B, 3B, 4B.

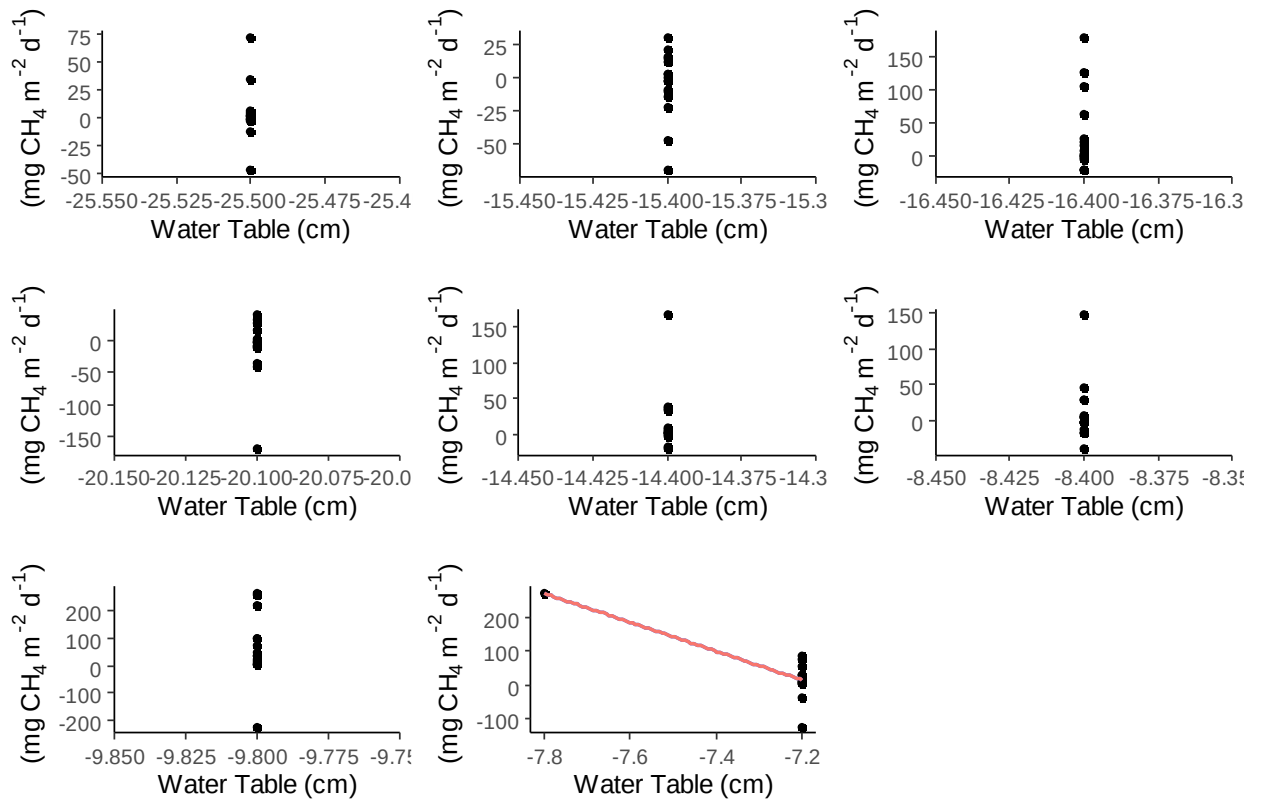


Figure A 13: Scatterplots depicting the variation in CH₄ fluxes across different water table depths (cm) at each of the Moderately Burned Fen's collars. The y-axis represents CH₄ fluxes, while the x-axis shows water table depths (cm). Collars are ordered as such in the figure: top left corner to bottom centre; 1A, 2A, 3A, 4A, 1B, 2B, 3B, 4B.

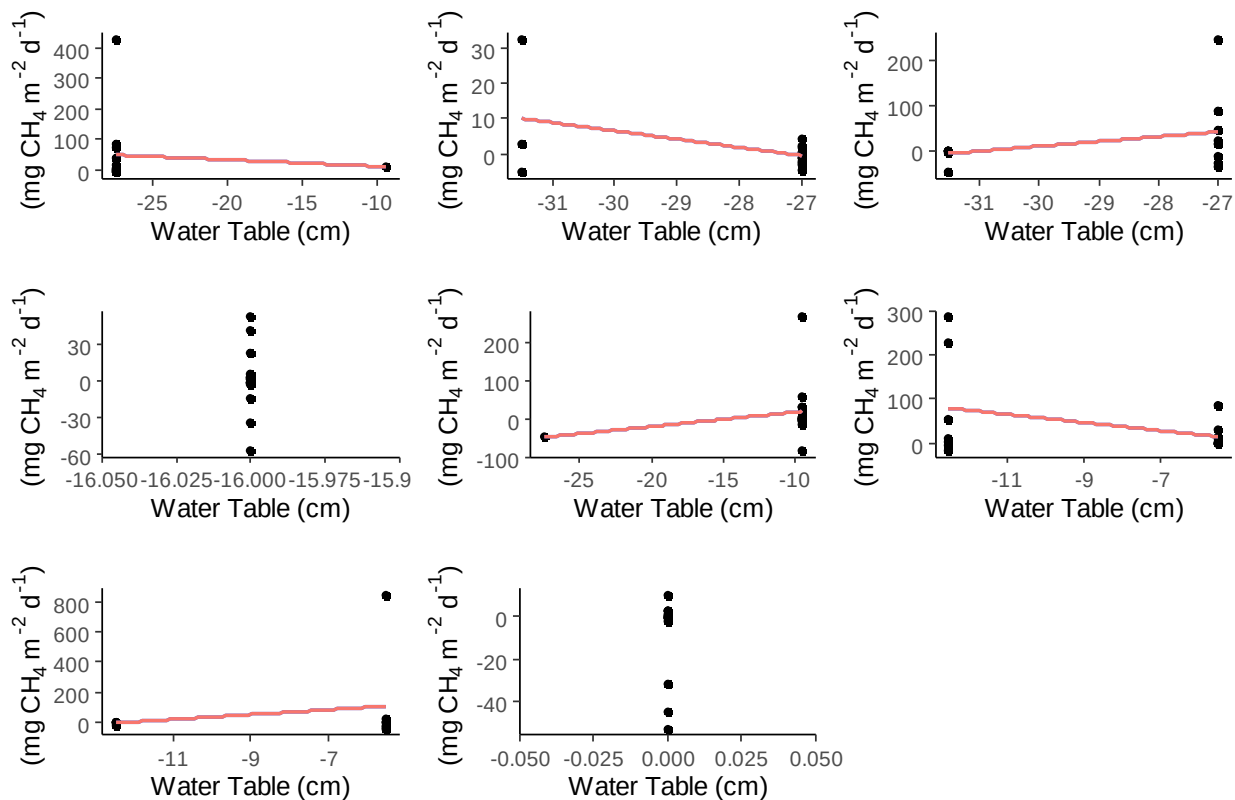


Figure A 14: Scatterplots depicting the variation in CH₄ fluxes across different water table depths (cm) at each of the Severely Burned Fen's collars. The y-axis represents CH₄ fluxes, while the x-axis shows water table depths (cm). Collars are ordered as such in the figure: top left corner to bottom centre; 1A, 2A, 3A, 4A, 1B, 2B, 3B, 4B.

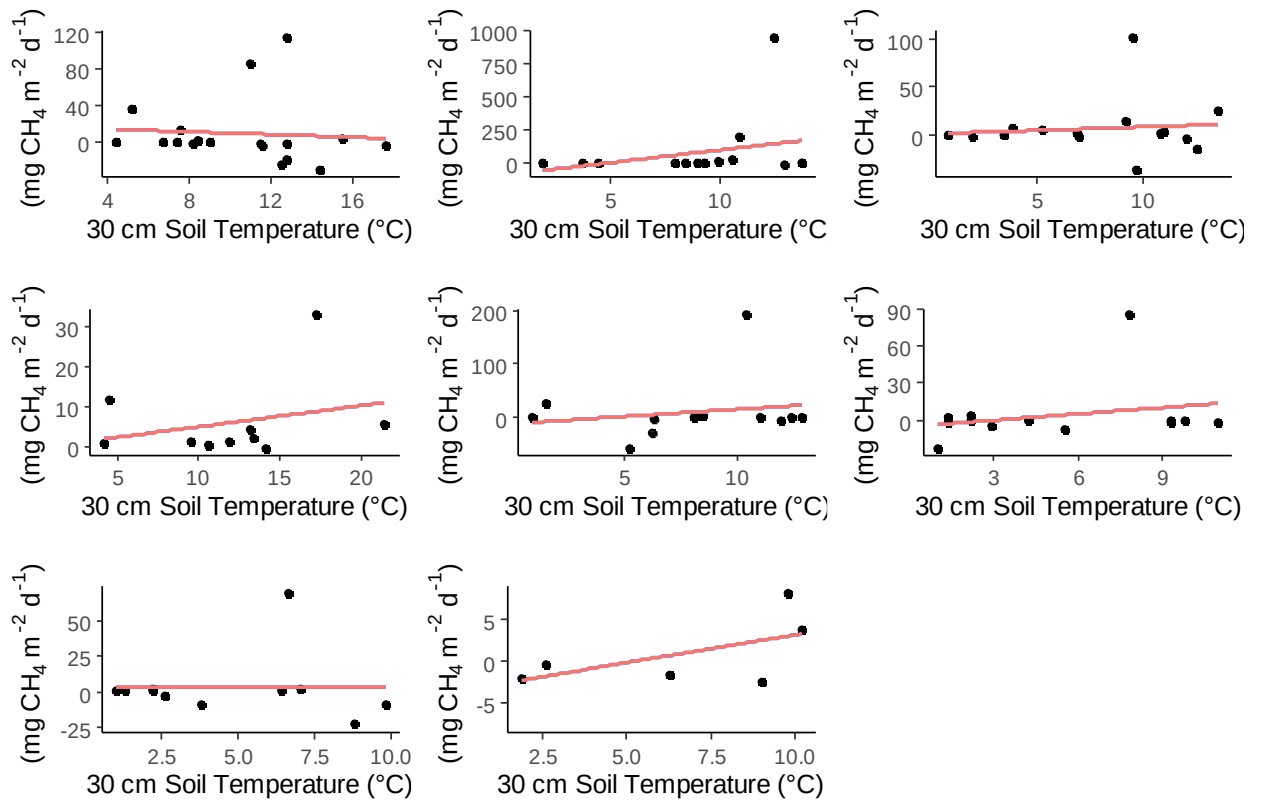


Figure A 15: Scatterplots depicting the variation in CH₄ fluxes across different 30 cm soil temperatures (°C) at each of the Unburned Bog's collars. The y-axis represents CH₄ fluxes, while the x-axis shows 30 cm soil temperatures (°C). Collars are ordered as such in the figure: top left corner to bottom centre; 1A, 2A, 3A, 4A, 1B, 2B, 3B, 4B.

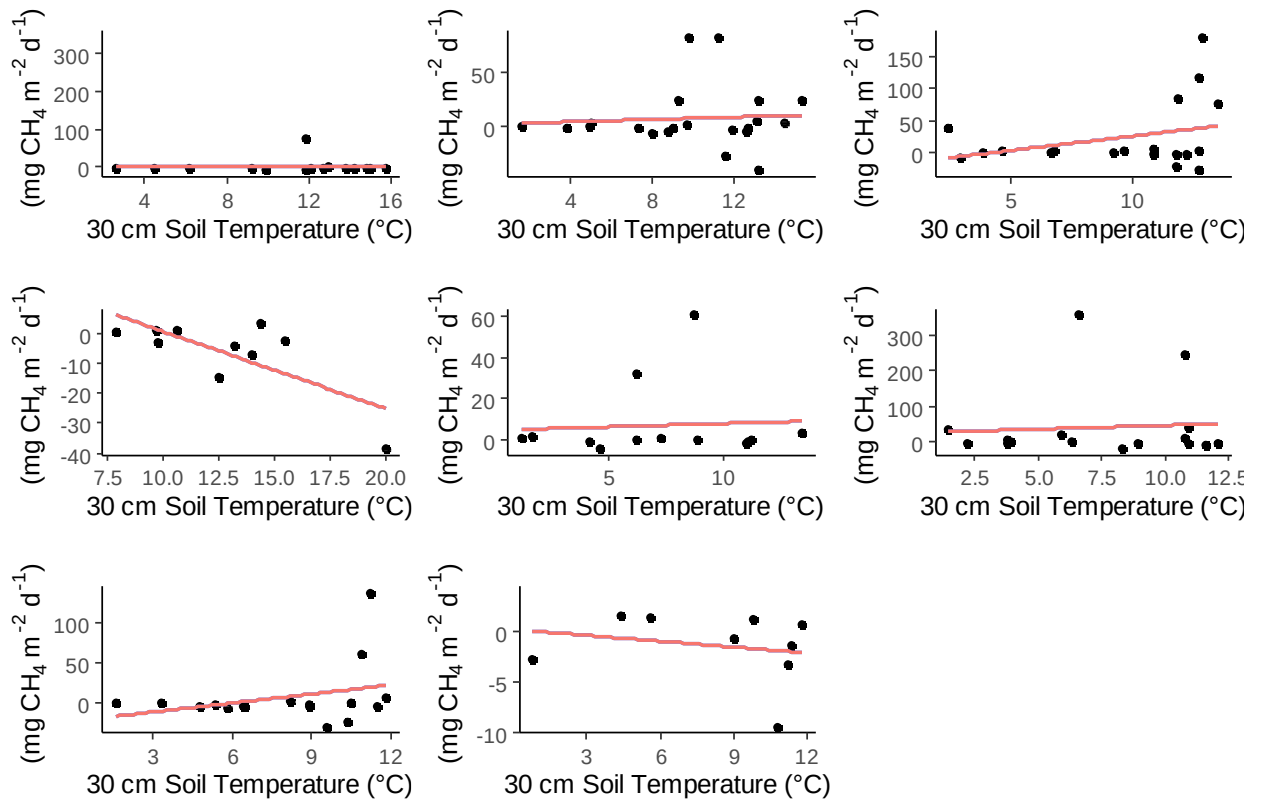


Figure A 16: Scatterplots depicting the variation in CH₄ fluxes across different 30 cm soil temperatures (°C) at each of the Moderately Burned Bog's collars. The y-axis represents CH₄ fluxes, while the x-axis shows 30 cm soil temperatures (°C). Collars are ordered as such in the figure: top left corner to bottom centre; 1A, 2A, 3A, 4A, 1B, 2B, 3B, 4B.

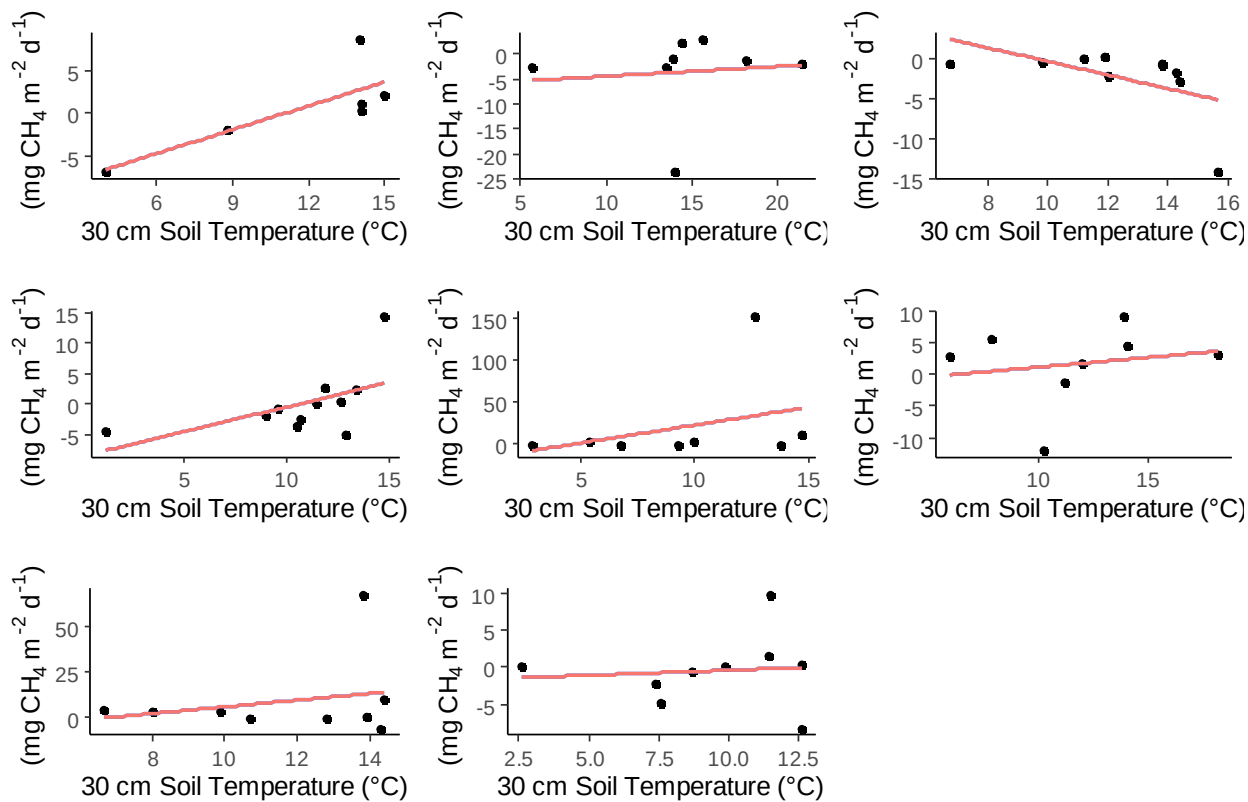


Figure A 17: Scatterplots depicting the variation in CH₄ fluxes across different 30 cm soil temperatures (°C) at each of the Severely Burned Bog's collars. The y-axis represents CH₄ fluxes, while the x-axis shows 30 cm soil temperatures (°C). Collars are ordered as such in the figure: top left corner to bottom centre; 1A, 2A, 3A, 4A, 1B, 2B, 3B, 4B.

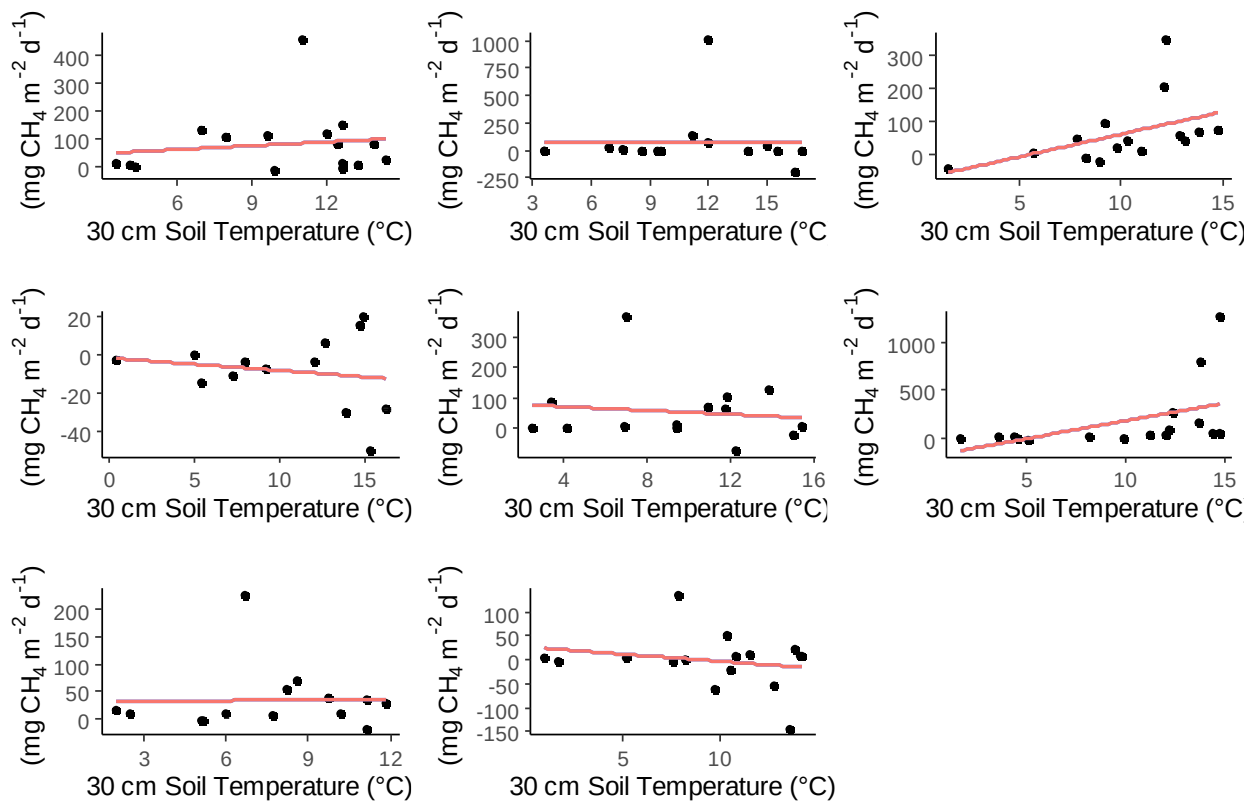


Figure A 18: Scatterplots depicting the variation in CH₄ fluxes across different 30 cm soil temperatures (°C) at each of the Unburned Fen's collars. The y-axis represents CH₄ fluxes, while the x-axis shows 30 cm soil temperatures (°C). Collars are ordered as such in the figure: top left corner to bottom centre; 1A, 2A, 3A, 4A, 1B, 2B, 3B, 4B.

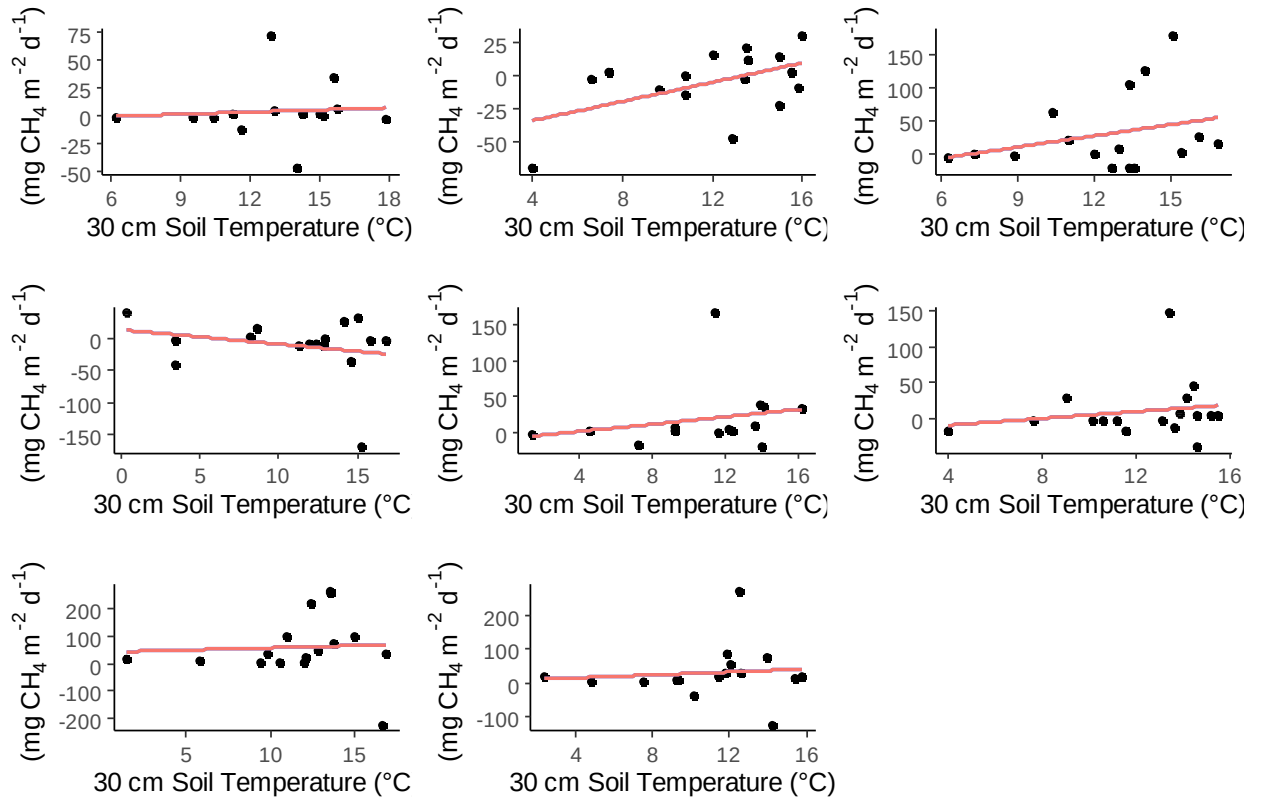


Figure A 19: Scatterplots depicting the variation in CH₄ fluxes across different 30 cm soil temperatures (°C) at each of the Moderately Burned Fen's collars. The y-axis represents CH₄ fluxes, while the x-axis shows 30 cm soil temperatures (°C). Collars are ordered as such in the figure: top left corner to bottom centre; 1A, 2A, 3A, 4A, 1B, 2B, 3B, 4B.

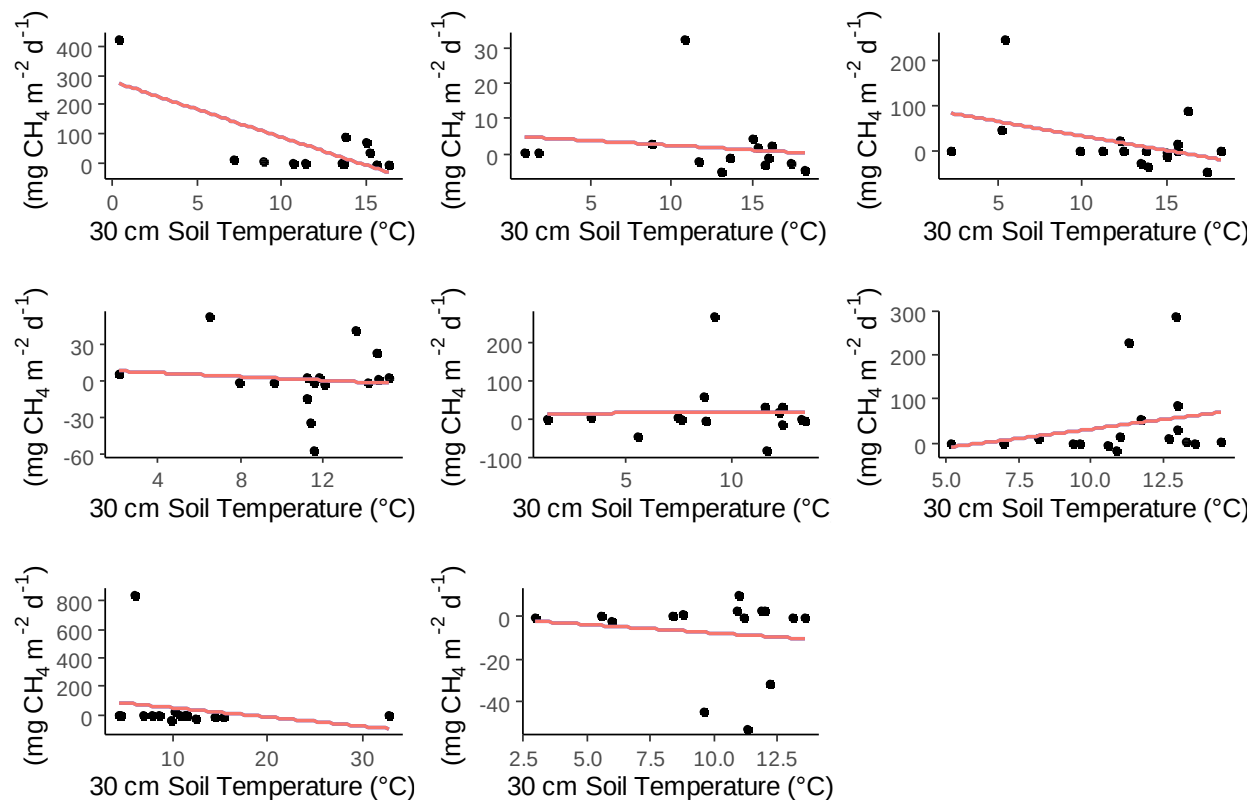


Figure A 20: Scatterplots depicting the variation in CH₄ fluxes across different 30 cm soil temperatures (°C) at each of the Severely Burned Fen’s collars. The y-axis represents CH₄ fluxes, while the x-axis shows 30 cm soil temperatures (°C). Collars are ordered as such in the figure: top left corner to bottom centre; 1A, 2A, 3A, 4A, 1B, 2B, 3B, 4B.

Table A 1: Pairwise comparisons of Plant Functional Traits, Biomass, and Vegetation Community Composition.

	Df	Sum Sq	Mean Sq	F-value	p-value
--	----	--------	---------	---------	---------

R. groenlandicum Height					
Severely Burned Fen – Moderately Burned Fen	15.333	7.25	13	2.115	0.2709
Severely Burned Fen – Unburned Fen	3.967	7.25	13	0.547	0.9804
Severely Burned Fen – Moderately Burned Bog	2.333	7.25	13	0.322	0.9974
Severely Burned Fen – Unburned Bog	0.333	6.28	13	0.053	1.0000
Moderately Burned Fen – Unburned Fen	-11.36	7.25	13	-1.568	0.5408
Moderately Burned Fen – Moderately Burned Bog	-13.00	7.25	13	-1.793	0.4173
Moderately Burned Fen – Unburned Bog	-15.00	6.28	13	-2.389	0.1793
Unburned Fen – Moderately Burned Bog	-1.633	7.25	13	-0.225	0.9993
Unburned Fen – Unburned Bog	-3.633	6.28	13	-0.579	0.9760
Moderately Burned Bog – Unburned Bog	-2.000	6.28	13	-0.319	0.9975
R. groenlandicum Leaf Dry Matter Content					
Severely Burned Fen – Moderately Burned Fen	0.0347	0.126	13	0.276	0.9986
Severely Burned Fen – Unburned Fen	-0.013	0.126	13	-0.108	1.0000
Severely Burned Fen – Moderately Burned Bog	-0.215	0.126	13	-1.712	0.4601
Severely Burned Fen – Unburned Bog	0.0235	0.109	13	0.216	0.9995
Moderately Burned Fen – Unburned Fen	-0.048	0.126	13	-0.383	0.9948
Moderately Burned Fen – Moderately Burned Bog	-0.250	0.126	13	-1.988	0.3238
Moderately Burned Fen – Unburned Bog	-0.011	0.109	13	-0.103	1.0000
Unburned Fen – Moderately Burned Bog	-0.201	0.126	13	-1.604	0.5199
Unburned Fen – Unburned Bog	0.0371	0.109	13	0.340	0.9967
Moderately Burned Bog – Unburned Bog	0.2388	0.109	13	2.193	0.2418
R. groenlandicum Specific Leaf Area					
Severely Burned Fen – Moderately Burned Fen	-1.198	1.95	13	-0.615	0.9701
Severely Burned Fen – Unburned Fen	-2.462	1.95	13	-1.263	0.7166
Severely Burned Fen – Moderately Burned Bog	1.197	1.95	13	0.614	0.9702
Severely Burned Fen – Unburned Bog	-2.852	1.69	13	-1.690	0.4723
Moderately Burned Fen – Unburned Fen	-1.264	1.95	13	-0.649	0.9639
Moderately Burned Fen – Moderately Burned Bog	2.396	1.95	13	1.229	0.7355
Moderately Burned Fen – Unburned Bog	-1.653	1.69	13	-0.980	0.8597
Unburned Fen – Moderately Burned Bog	3.660	1.95	13	1.878	0.3748
Unburned Fen – Unburned Bog	-0.389	1.69	13	-0.231	0.9993
Moderately Burned Bog – Unburned Bog	-4.049	1.69	13	-2.399	0.1765
P. mariana height					
Severely Burned Fen – Moderately Burned Fen	-10.23	2.68	9	-3.815	0.0259
Severely Burned Fen – Unburned Fen	0.40	2.68	9	0.149	0.9999

Severely Burned Fen – Moderately Burned Bog	-2.50	3.00	9	-0.834	0.9135
Severely Burned Fen – Unburned Bog	3.00	2.68	9	1.118	0.7937
Moderately Burned Fen – Unburned Fen	10.63	2.68	9	3.964	0.0209
Moderately Burned Fen – Moderately Burned Bog	7.73	3.00	9	2.578	0.1558
Moderately Burned Fen – Unburned Bog	13.23	2.68	9	4.933	0.0055
Unburned Fen – Moderately Burned Bog	-2.90	3.00	9	-0.967	0.8633
Unburned Fen – Unburned Bog	2.60	2.68	9	0.969	0.8623
Moderately Burned Bog – Unburned Bog	5.50	3.00	9	1.834	0.4122
P. mariana Specific Leaf Area					
Severely Burned Fen – Moderately Burned Fen	3.68	2.38	8	1.546	0.5647
Severely Burned Fen – Unburned Fen	9.09	2.38	8	3.814	0.0308
Severely Burned Fen – Moderately Burned Bog	4.81	2.61	8	1.844	0.4135
Severely Burned Fen – Unburned Bog	7.29	2.38	8	3.060	0.0857
Moderately Burned Fen – Unburned Fen	5.40	2.13	8	2.537	0.1743
Moderately Burned Fen – Moderately Burned Bog	1.13	2.38	8	0.474	0.9877
Moderately Burned Fen – Unburned Bog	3.61	2.13	8	1.693	0.4869
Unburned Fen – Moderately Burned Bog	-4.27	2.38	8	-1.795	0.4367
Unburned Fen – Unburned Bog	-1.80	2.13	8	-0.843	0.9095
Moderately Burned Bog – Unburned Bog	2.48	2.38	8	1.040	0.8305
P. mariana Leaf Dry Matter Content					
Severely Burned Fen – Moderately Burned Fen	0.1303	0.144	10	0.908	0.8875
Severely Burned Fen – Unburned Fen	-0.453	0.144	10	-3.163	0.0609
Severely Burned Fen – Moderately Burned Bog	-0.166	0.144	10	-1.160	0.7724
Severely Burned Fen – Unburned Bog	-0.412	0.144	10	-2.874	0.0951
Moderately Burned Fen – Unburned Fen	-0.584	0.144	10	-4.071	0.0150
Moderately Burned Fen – Moderately Burned Bog	-0.296	0.144	10	-2.069	0.3034
Moderately Burned Fen – Unburned Bog	-0.542	0.144	10	-3.782	0.0234
Unburned Fen – Moderately Burned Bog	0.2874	0.144	10	2.003	0.3304
Unburned Fen – Unburned Bog	0.0415	0.144	10	0.289	0.9982
Moderately Burned Bog – Unburned Bog	-0.245	0.144	10	-1.713	0.4680
B. glandulosa Height					
Severely Burned Fen – Moderately Burned Fen	-90.5	20.7	6	-4.381	0.0111
Severely Burned Fen – Unburned Fen	15.2	20.7	6	0.737	0.7519
Moderately Burned Fen – Unburned Fen	105.8	20.7	6	5.118	0.0052
B. glandulosa Specific Leaf Area					
Severely Burned Fen – Moderately Burned Fen	-2.278	0.935	6	-2.437	0.1106

Severely Burned Fen – Unburned Fen	-2.049	0.935	6	-2.192	0.1513
Moderately Burned Fen – Unburned Fen	0.229	0.935	6	0.245	0.9675
B. glandulosa Leaf Dry Matter Content					
Severely Burned Fen – Moderately Burned Fen	0.0170	0.0403	6	0.422	0.9081
Severely Burned Fen – Unburned Fen	-0.090	0.0403	6	-2.243	0.1417
Moderately Burned Fen – Unburned Fen	-0.107	0.0403	6	-2.665	0.0828
E. fluviatile Height					
Severely Burned Fen – Moderately Burned Fen	-23.03	6.93	5	-3.324	0.0464
Severely Burned Fen – Unburned Fen	-0.85	7.75	5	-0.110	0.9934
Moderately Burned Fen – Unburned Fen	22.18	7.75	5	2.863	0.0767
E. fluviatile Specific Leaf Area					
Severely Burned Fen – Moderately Burned Fen	4.268	1.63	6	2.617	0.0880
Severely Burned Fen – Unburned Fen	4.516	1.63	6	2.769	0.0726
Moderately Burned Fen – Unburned Fen	0.248	1.63	6	0.152	0.9874
E. fluviatile Leaf Dry Matter Content					
Severely Burned Fen – Moderately Burned Fen	0.035	0.0308	6	1.134	0.5299
Severely Burned Fen – Unburned Fen	-0.078	0.0308	6	-2.531	0.0982
Moderately Burned Fen – Unburned Fen	-0.113	0.0308	6	-3.665	0.0245
V. vitis-ideae Height					
Moderately Burned Bog – Unburned Bog	-7	1.33	4	-5.248	0.0063
V. vitis-ideae Specific Leaf Area					
Moderately Burned Bog – Unburned Bog	-0.093	1.23	4	-0.076	0.9430
V. vitis-ideae Leaf Dry Matter Content					
Moderately Burned Bog – Unburned Bog	-0.003	0.0391	4	-0.084	0.9371
E. sylvaticum Height					
Moderately Burned Bog – Unburned Bog	-6.8	5.31	4	-1.280	0.2697
E. sylvaticum Specific Leaf Area					
Moderately Burned Bog – Unburned Bog	0.78	0.989	4	0.789	0.4744
E. sylvaticum Leaf Dry Matter Content					
Moderately Burned Bog – Unburned Bog	0.0801	0.0322	4	2.487	0.0677
E. arvense Height					
Moderately Burned Bog – Unburned Bog	-2.67	2.15	4	-1.241	0.2824
E. arvense Specific Leaf Area					
Moderately Burned Bog – Unburned Bog	0.971	0.87	4	1.116	0.3269
E. arvense Leaf Dry Matter Content					
Moderately Burned Bog – Unburned Bog	-0.015	0.0085	4	-1.784	0.1491
Biomass					
Moderately Burned Bog - Severely Burned Bog	0.914	4.55	7280	0.201	1.0000
Moderately Burned Bog – Unburned Bog	-3.145	4.75	7280	-0.661	0.9860

Moderately Burned Bog – Moderately Burned Fen	-7.177	4.54	7280	-1.581	0.6112
Moderately Burned Bog – Severely Burned Fen	-0.472	4.25	7280	-0.111	1.0000
Moderately Burned Bog – Unburned Fen	-7.800	5.31	7280	-1.469	0.6842
Severely Burned Bog – Unburned Bog	-4.059	3.78	7280	-1.075	0.8915
Severely Burned Bog – Moderately Burned Fen	-8.091	3.50	7280	-2.311	0.1897
Severely Burned Bog – Severely Burned Fen	-1.386	3.12	7280	-0.444	0.9978
Severely Burned Bog – Unburned Fen	-8.714	4.46	7280	-1.956	0.3681
Unburned Bog – Moderately Burned Fen	-4.032	3.76	7280	-1.072	0.8928
Unburned Bog – Severely Burned Fen	2.673	3.41	7280	0.784	0.9703
Unburned Bog – Unburned Fen	-4.655	4.66	7280	-0.998	0.9187
Moderately Burned Fen – Severely Burned Fen	6.705	3.10	7280	2.161	0.2565
Moderately Burned Fen – Unburned Fen	-0.623	4.44	7280	-0.140	1.0000
Severely Burned Fen – Unburned Fen	-7.328	4.15	7280	-1.766	0.4880

Table A 2: Pairwise comparisons of percent cover by functional group

Shrub	estimate	SE	df	T ratio	p-value
Moderately Burned Bog – Severely Burned Bog	15.211	8.39	180	1.812	0.4604
Moderately Burned Bog – Unburned Bog	-9.304	5.73	180	-1.623	0.5845
Moderately Burned Bog – Moderately Burned Fen	-0.409	4.95	180	-0.083	1.0000

Moderately Burned Bog – Severely Burned Fen	1.140	5.47	180	0.208	0.9999
Moderately Burned Bog – Unburned Fen	5.369	4.79	180	1.121	0.8723
Severely Burned Bog – Unburned Bog	-24.516	8.39	180	-2.921	0.0448
Severely Burned Bog – Moderately Burned Fen	-15.620	7.88	180	-1.983	0.3561
Severely Burned Bog – Severely Burned Fen	-14.071	8.22	180	-1.713	0.5252
Severely Burned Bog – Unburned Fen	-9.842	7.78	180	-1.265	0.8035
Unburned Bog – Moderately Burned Fen	8.895	4.95	180	1.798	0.4696
Unburned Bog – Severely Burned Fen	10.444	5.47	180	1.909	0.4001
Unburned Bog – Unburned Fen	14.673	4.79	180	3.063	0.0299
Moderately Burned Fen – Severely Burned Fen	1.549	4.64	180	0.334	0.9994
Moderately Burned Fen – Unburned Fen	5.778	3.82	180	1.514	0.6556
Severely Burned Fen – Unburned Fen	4.229	4.47	180	0.945	0.9341
Leaf Litter					
Moderately Burned Bog – Severely Burned Bog	-0.667	10.89	105	-0.061	1.0000
Moderately Burned Bog – Unburned Bog	-7.667	6.93	105	-1.106	0.8777
Moderately Burned Bog – Moderately Burned Fen	-21.580	6.61	105	-3.267	0.0179
Moderately Burned Bog – Severely Burned Fen	-32.667	6.93	105	-4.714	0.0001
Moderately Burned Bog – Unburned Fen	-12.792	6.54	105	-1.956	0.3746
Severely Burned Bog – Unburned Bog	-7.000	11.00	105	-0.636	0.9880
Severely Burned Bog – Moderately Burned Fen	-20.913	10.80	105	-1.937	0.3860
Severely Burned Bog – Severely Burned Fen	-32.000	11.00	105	-2.909	0.0492
Severely Burned Bog – Unburned Fen	-12.125	10.76	105	-1.127	0.8691
Unburned Bog – Moderately Burned Fen	-13.913	6.78	105	-2.051	0.3214
Unburned Bog – Severely Burned Fen	-25.000	7.10	105	-3.521	0.0082
Unburned Bog – Unburned Fen	-5.125	6.72	105	-0.763	0.9731
Moderately Burned Fen – Severely Burned Fen	-11.087	6.78	105	-1.634	0.5781
Moderately Burned Fen – Unburned Fen	8.788	6.39	105	1.376	0.7411
Severely Burned Fen – Unburned Fen	19.875	6.72	105	2.957	0.0432
Moss					
Moderately Burned Bog – Severely Burned Bog	6.933	10.13	189	0.684	0.9835
Moderately Burned Bog – Unburned Bog	-9.317	8.18	189	-1.139	0.8646
Moderately Burned Bog – Moderately Burned Fen	3.489	7.63	189	0.457	0.9975
Moderately Burned Bog – Severely Burned Fen	14.999	7.46	189	2.011	0.3400
Moderately Burned Bog – Unburned Fen	-0.432	7.38	189	-0.059	1.0000
Severely Burned Bog – Unburned Bog	-16.250	10.35	189	-1.570	0.6193
Severely Burned Bog – Moderately Burned Fen	-3.444	9.92	189	-0.347	0.9993
Severely Burned Bog – Severely Burned Fen	8.066	9.79	189	0.824	0.9628
Severely Burned Bog – Unburned Fen	-7.365	9.73	189	-0.757	0.9742
Unburned Bog – Moderately Burned Fen	12.806	7.92	189	1.617	0.5885
Unburned Bog – Severely Burned Fen	24.316	7.75	189	3.137	0.0239
Unburned Bog – Unburned Fen	8.885	7.68	189	1.157	0.8565

Moderately Burned Fen – Severely Burned Fen	11.510	7.17	189	1.605	0.5961
Moderately Burned Fen – Unburned Fen	-3.921	7.09	189	-0.553	0.9938
Severely Burned Fen – Unburned Fen	-15.432	6.90	189	-2.236	0.2264
Forb					
Moderately Burned Bog – Severely Burned Bog	7.180	5.92	435	1.212	0.8310
Moderately Burned Bog – Unburned Bog	1.820	2.47	435	0.738	0.9771
Moderately Burned Bog – Moderately Burned Fen	13.405	2.59	435	5.175	<.0001
Moderately Burned Bog – Severely Burned Fen	3.879	2.47	435	1.570	0.6187
Moderately Burned Bog – Unburned Fen	8.146	2.57	435	3.172	0.0200
Severely Burned Bog – Unburned Bog	-5.360	5.84	435	-0.918	0.9417
Severely Burned Bog – Moderately Burned Fen	6.225	5.89	435	1.057	0.8981
Severely Burned Bog – Severely Burned Fen	-3.301	5.84	435	-0.565	0.9932
Severely Burned Bog – Unburned Fen	0.967	5.88	435	0.164	1.0000
Unburned Bog – Moderately Burned Fen	11.585	2.38	435	4.862	<.0001
Unburned Bog – Severely Burned Fen	2.059	2.25	435	0.914	0.9427
Unburned Bog – Unburned Fen	6.327	2.36	435	2.682	0.0809
Moderately Burned Fen – Severely Burned Fen	-9.526	2.39	435	-3.989	0.0011
Moderately Burned Fen – Unburned Fen	-5.258	2.49	435	-2.113	0.2826
Severely Burned Fen – Unburned Fen	4.267	2.36	435	1.805	0.4635

Table A 3: Pairwise comparisons for CO₂, CH₄, GCC, and environmental variables

	estimate	SE	df	t.ratio	p-value
5 cm Temperature					
Moderately Burned Bog – Severely Burned Bog	-1.5796	1.138	62	-1.389	0.7336
Moderately Burned Bog – Unburned Bog	-0.3553	0.913	62	-0.389	0.9988
Moderately Burned Bog – Moderately Burned Fen	-0.6130	0.827	62	-0.741	0.9759

Moderately Burned Bog – Severely Burned Fen	-3.3310	0.834	62	-3.992	0.0023
Moderately Burned Bog – Unburned Fen	-0.3288	0.834	62	-0.395	0.9987
Severely Burned Bog – Unburned Bog	1.2243	1.138	62	1.076	0.8890
Severely Burned Bog – Moderately Burned Fen	0.9665	1.070	62	0.903	0.9442
Severely Burned Bog – Severely Burned Fen	-1.7514	1.075	62	-1.629	0.5832
Severely Burned Bog – Unburned Fen	1.2507	1.075	62	1.164	0.8521
Unburned Bog – Moderately Burned Fen	-0.2578	0.828	62	-0.311	0.9996
Unburned Bog – Severely Burned Fen	-2.9757	0.835	62	-3.565	0.0089
Unburned Bog – Unburned Fen	0.0265	0.834	62	0.032	1.0000
Moderately Burned Fen – Severely Burned Fen	-2.7179	0.740	62	-3.675	0.0063
Moderately Burned Fen – Unburned Fen	0.2842	0.739	62	0.385	0.9989
Severely Burned Fen – Unburned Fen	3.0021	0.746	62	4.022	0.0021
30 cm Soil Temperature					
Moderately Burned Bog – Severely Burned Bog	-3.369	0.732	62	-4.600	0.0003
Moderately Burned Bog – Unburned Bog	0.887	0.590	62	1.504	0.6631
Moderately Burned Bog – Moderately Burned Fen	-1.946	0.512	62	-3.799	0.0043
Moderately Burned Bog – Severely Burned Fen	-1.554	0.519	62	-2.993	0.0438
Moderately Burned Bog – Unburned Fen	-0.131	0.522	62	-0.251	0.9999
Severely Burned Bog – Unburned Bog	4.256	0.739	62	5.759	<.0001
Severely Burned Bog – Moderately Burned Fen	1.423	0.679	62	2.097	0.3025
Severely Burned Bog – Severely Burned Fen	1.815	0.684	62	2.654	0.0997
Severely Burned Bog – Unburned Fen	3.238	0.686	62	4.719	0.0002
Unburned Bog – Moderately Burned Fen	-2.833	0.522	62	-5.430	<.0001
Unburned Bog – Severely Burned Fen	-2.441	0.529	62	-4.618	0.0003
Unburned Bog – Unburned Fen	-1.018	0.531	62	-1.915	0.4029
Moderately Burned Fen – Severely Burned Fen	0.392	0.440	62	0.891	0.9473
Moderately Burned Fen – Unburned Fen	1.815	0.444	62	4.091	0.0017
Severely Burned Fen – Unburned Fen	1.423	0.452	62	3.150	0.0289
GEP					
Moderately Burned Bog – Severely Burned Bog	2.69	2.57	62	1.046	0.9001
Moderately Burned Bog – Unburned Bog	1.23	2.22	62	0.551	0.9937
Moderately Burned Bog – Moderately Burned Fen	6.82	2.19	62	3.117	0.0316
Moderately Burned Bog – Severely Burned Fen	13.08	2.18	62	5.992	<.0001
Moderately Burned Bog – Unburned Fen	5.80	2.16	62	2.683	0.0932
Severely Burned Bog – Unburned Bog	-1.46	2.57	62	-0.570	0.9926
Severely Burned Bog – Moderately Burned Fen	4.13	2.54	62	1.628	0.5837

Severely Burned Bog – Severely Burned Fen	10.39	2.53	62	4.104	0.0016
Severely Burned Bog – Unburned Fen	3.11	2.51	62	1.237	0.8172
Unburned Bog – Moderately Burned Fen	5.59	2.19	62	2.559	0.1232
Unburned Bog – Severely Burned Fen	11.85	2.18	62	5.437	<.0001
Unburned Bog – Unburned Fen	4.57	2.16	62	2.118	0.2915
Moderately Burned Fen – Severely Burned Fen	6.26	2.14	62	2.923	0.0523
Moderately Burned Fen – Unburned Fen	-1.02	2.12	62	-0.482	0.9966
Severely Burned Fen – Unburned Fen	-7.28	2.11	62	-3.445	0.0126
ER					
Moderately Burned Bog – Severely Burned Bog	-1.699	1.91	62	-0.889	0.9478
Moderately Burned Bog – Unburned Bog	-1.508	1.65	62	-0.912	0.9421
Moderately Burned Bog – Moderately Burned Fen	-6.178	1.63	62	-3.792	0.0044
Moderately Burned Bog – Severely Burned Fen	-9.683	1.63	62	-5.957	<.0001
Moderately Burned Bog – Unburned Fen	-6.854	1.61	62	-4.262	0.0010
Severely Burned Bog – Unburned Bog	0.191	1.91	62	0.100	1.0000
Severely Burned Bog – Moderately Burned Fen	-4.479	1.89	62	-2.373	0.1817
Severely Burned Bog – Severely Burned Fen	-7.983	1.88	62	-4.238	0.0010
Severely Burned Bog – Unburned Fen	-5.154	1.87	62	-2.758	0.0783
Unburned Bog – Moderately Burned Fen	-4.670	1.63	62	-2.871	0.0595
Unburned Bog – Severely Burned Fen	-8.174	1.62	62	-5.037	0.0001
Unburned Bog – Unburned Fen	-5.345	1.61	62	-3.329	0.0176
Moderately Burned Fen – Severely Burned Fen	-3.505	1.60	62	-2.195	0.2549
Moderately Burned Fen – Unburned Fen	-0.675	1.58	62	-0.428	0.9981
Severely Burned Fen – Unburned Fen	2.829	1.58	62	1.796	0.4757
ER & Environmental Variables					
Moderately Burned Bog – Severely Burned Bog	-1.843	1.72	69	-1.074	0.8901
Moderately Burned Bog – Unburned Bog	-1.706	1.54	69	-1.110	0.8757
Moderately Burned Bog – Moderately Burned Fen	-6.100	1.59	69	-3.832	0.0036
Moderately Burned Bog – Severely Burned Fen	-9.521	1.60	69	-5.954	<.0001
Moderately Burned Bog – Unburned Fen	-6.701	1.54	69	-4.346	0.0006
Severely Burned Bog – Unburned Bog	0.138	1.71	69	0.081	1.0000
Severely Burned Bog – Moderately Burned Fen	-4.257	1.76	69	-2.422	0.1630
Severely Burned Bog – Severely Burned Fen	-7.678	1.76	69	-4.352	0.0006
Severely Burned Bog – Unburned Fen	-4.858	1.71	69	-2.837	0.0634
Unburned Bog – Moderately Burned Fen	-4.394	1.58	69	-2.778	0.0733
Unburned Bog – Severely Burned Fen	-7.816	1.59	69	-4.917	0.0001

Unburned Bog – Unburned Fen	-4.996	1.53	69	-3.261	0.0205
Moderately Burned Fen – Severely Burned Fen	-3.421	1.64	69	-2.083	0.3083
Moderately Burned Fen – Unburned Fen	-0.601	1.59	69	-0.379	0.9989
Severely Burned Fen – Unburned Fen	2.820	1.59	69	1.769	0.4924
GCC					
Moderately Burned Bog – Severely Burned Bog	0.00168	0.00757	36	0.222	0.9999
Moderately Burned Bog – Unburned Bog	-0.01909	0.00745	36	-2.563	0.1332
Moderately Burned Bog – Moderately Burned Fen	-0.03896	0.00740	36	-5.263	0.0001
Moderately Burned Bog – Severely Burned Fen	-0.04203	0.00739	36	-5.685	<.0001
Moderately Burned Bog – Unburned Fen	-0.04065	0.00739	36	-5.498	<.0001
Severely Burned Bog – Unburned Bog	-0.02077	0.00757	36	-2.743	0.0911
Severely Burned Bog – Moderately Burned Fen	-0.04063	0.00752	36	-5.401	0.0001
Severely Burned Bog – Severely Burned Fen	-0.04371	0.00752	36	-5.816	<.0001
Severely Burned Bog – Unburned Fen	-0.04233	0.00752	36	-5.632	<.0001
Unburned Bog – Moderately Burned Fen	-0.01986	0.00740	36	-2.685	0.1033
Unburned Bog – Severely Burned Fen	-0.02294	0.00739	36	-3.104	0.0398
Unburned Bog – Unburned Fen	-0.02156	0.00739	36	-2.917	0.0618
Moderately Burned Fen – Severely Burned Fen	-0.00308	0.00734	36	-0.419	0.9982
Moderately Burned Fen – Unburned Fen	-0.00169	0.00734	36	-0.231	0.9999
Severely Burned Fen – Unburned Fen	0.00139	0.00733	36	0.189	1.0000

Table A 4: Marginal and Conditional R² values

	Marginal R ²	Conditional R ²
Water Table	0.23	0.57
5 cm Soil Temperature	0.09	0.10
30 cm Soil Temperature	0.15	0.15
NEE	0.08	0.30
GEP	0.15	0.26
ER	0.15	0.26

NEE and Environmental Variables	0.06	0.29
GEP and Environmental Variables	0.22	0.32
ER and Environmental Variables	0.18	0.33
CH ₄	0.22	0.32
CH ₄ and Environmental Variables	0.03	0.60

Table A 5: Incremental R² values

	Incremental R ² for Site	Incremental R ² for Microform	Incremental R ² for 5 cm Soil Temperature	Incremental R ² for 30 cm Soil Temperature	Incremental R ² for Water Table
Water Table	0.05	0.17			
5 cm Soil Temperature	0.04	0.05			
30 cm Soil Temperature	0.09	0.07			
NEE	0.03	0.02			
GEP	0.14	0.01			
ER	0.14	0.00			
NEE and Environmental Variables	0.03		0.00		5.56 e-05
GEP and Environmental Variables	0.14		0.02		0.00
ER and Environmental Variables	0.14		0.02		0.00
CH ₄	0.03	0.00			
CH ₄ and Environmental Variables	0.03			0.00	0.00

Table A 6: Semi-partial R² values

	Semi-partial R ² for Microform	Semi-partial R ² for Site	Semi-partial R ² for 5 cm Soil Temperature	Semi-partial R ² for 30 cm Soil Temperature	Semi-partial R ² for Water Table
Water Table	0.19	0.06			
5 cm Soil Temperature	0.04	0.03			

30 cm Soil Temperature	0.04	0.06			
NEE	0.01	0.03			
GEP	0.00	0.13			
ER	0.14	-3.21e-05			
NEE and Environmental Variables		0.00	0.03		0.03
GEP and Environmental Variables		0.01	0.12		0.15
ER and Environmental Variables		0.01	0.13		0.15
CH4	0.00	0.03			
CH4 and Environmental Variables		0.00		0.03	0.03

Table A 7: GEP and environmental variables model with 30 cm soil temperature

GEP & Environmental Variables				
Intercept	1	692	7.844485	0.0052
30 cm Soil Temperature	1	629	0.873951	0.3502
Water Table	1	629	0.000022	0.9962
Site	5	68	1.157611	0.3391
30 cm Soil Temperature * Water Table	1	629	0.273491	0.6012
30 cm Soil Temperature * Site	5	629	0.834695	0.5252
Water Table * Site	5	629	2.685358	0.0205
30 cm Soil Temperature * Water Table * Site	5	629	5.013399	0.0002